PROGRAMA DE DOCTORADO EN CIENCIAS ECONÓMICAS, EMPRESARIALES Y JURÍDICAS

TESIS DOCTORAL

# VALORACIÓN DE SERVICIOS Y CONTRASERVICIOS DE LOS AGROECOSISTEMAS: UN ENFOQUE DE NO MERCADO PARA LOS AGROECOSISTEMAS SEMIÁRIDOS MEDITERRÁNEOS

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por la Universidad Politécnica de Cartagena

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PhD THESIS

# AGROECOSYSTEM SERVICES AND DISSERVICES VALUATION: A NON-MARKET APPROACH FOR SEMIARID MEDITERRANEAN AGROECOSYSTEMS

Presented by José Ángel Zabala García
to the Technical University of Cartagena in fulfilment of
the thesis requirement for the award of PhD

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*To all those who once trusted me*

## Preface and acknowledgements

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Without them, all this would not have been possible.

What follows is for them.

José A. Zabala

## Abstract

The contribution of agriculture to human wellbeing goes beyond food production. It also encompasses the provision of non-commodity goods and services that may impact human wellbeing both positively and negatively. Agriculture is the main activity developed within agroecosystems, where human pressures, mainly through agricultural practices, affect their innate functioning. This leads to the provision of agroecosystem services and disservices. Economic valuation of agroecosystem services and disservices, and agricultural practices in accordance, allow us to guide policy decisions in line with the contribution of agroecosystems to human wellbeing. In such a context, this thesis aims to economically evaluate the integrated social demand of agroecosystem services and disservices, and the agricultural practices that promotes them, by adapting a comprehensive approach for agroecosystem valuation in a semiarid Mediterranean region – the Region of Murcia (SE Spain). Discrete choice experiment was the central methodology used. The non-market value of agroecosystem services and disservices, and consequently, of agricultural practices impacting on them, was disentangled aiming to reflect the contribution of agroecosystems to human wellbeing in monetary terms. The validation of a comprehensive approach for agroecosystem valuation by stakeholders settled the agroecosystem services and disservices whose social demand and non-market value was then estimated. Regardless of the specific economic values, the non-market results provide deep insight into the expected focus of agricultural policies for increasing their impact on human wellbeing. Agricultural measures should therefore be centred around increasing food provision, promoting agroecosystem biodiversity, reducing the local temperature, generating opportunities for recreation in agricultural landscapes, whilst seeking to regulate water supply for irrigation and mitigate agricultural nutrient pollution. In accordance with the latter, the social demand for agricultural measures aiming to reduce nitrate pollution from agriculture was addressed. Despite the great preference heterogeneity, social support was revealed to the measures, providing guidance for policy makers in the establishment of socially supported strategies for agricultural nitrate pollution mitigation. To conclude, this thesis expects to provide better insight into the links between agriculture and human wellbeing, in the hopes that better- informed policy actions will, therefore, be developed that aim to boost human wellbeing.

**Keywords**: Agriculture; Discrete choice experiment; Environmental economics; Human wellbeing; Non-market valuation; Preference heterogeneity.

## Resumen

La contribución de la agricultura al bienestar humano va más allá de la mera producción de alimentos y materias primas. También abarca la provisión de bienes y servicios de no mercado que pueden tener impacto, positivo o negativo, en el bienestar humano. Así, la agricultura es la principal actividad desarrollada al amparo de los agroecosistemas, donde la presión antrópica de las prácticas agrícolas afecta a su funcionamiento y a los niveles de provisión de servicios y contraservicios ecosistémicos. La valoración económica de estos servicios y contraservicios, y en consecuencia de las prácticas agrícolas que los fomentan, permite orientar las decisiones de política agrícola de acuerdo a la contribución de los agroecosistemas al bienestar humano. En este contexto, la presente tesis doctoral tiene como objetivo central la valoración económica de la demanda social de servicios y contraservicios de los agroecosistemas, así como de las prácticas agrícolas que los promueven. Para ello, y utilizando experimentos de elección como metodología principal, se formula un enfoque integral para la valoración de agroecosistemas en una región semiárida del Mediterráneo, la Región de Murcia (sudeste de España). La validación de este enfoque es llevada a cabo por parte de los agentes implicados en su gestión y permite seleccionar los servicios y contraservicios más importantes, cuya demanda social y valor de no mercado son posteriormente estimados. Más allá de las cifras concretas, los resultados ofrecen una visión amplia sobre el enfoque necesario en las políticas agrícolas para que logren aumentar el bienestar de la sociedad. Así, las acciones de política agrícola deberían centrarse en aumentar la provisión de alimentos, promover la biodiversidad de los agroecosistemas, reducir la temperatura local, generar oportunidades para el ocio y recreo en los paisajes agrícolas, mientras se busca regular el suministro de agua para riego y mitigar la contaminación por nutrientes agrícolas. En relación a este último aspecto, en esta tesis también se aborda la evaluación de la demanda social de medidas agrícolas con el fin de reducir la contaminación por nitratos de la agricultura. Todas las medidas propuestas, pese a la heterogeneidad detectada en su demanda, contaron con apoyo social, lo que ha de brindar orientación a los gestores públicos para el establecimiento de estrategias para la mitigación de la contaminación agrícola por nitratos que sean socialmente respaldadas. Así se espera que esta tesis proporcione una mejor comprensión de los vínculos entre agricultura y bienestar humano, orientando la formulación de políticas mejor informadas y eficaces en aras de su contribución al bienestar social.

**Palabras clave**: Agricultura; Bienestar humano; Economía ambiental; Experimentos de elección; Heterogeneidad social; Valoración de no mercado.

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# Chapter 1. General introduction and objectives

## 1.1. Thesis focus

This PhD thesis is based on three main and interlinked notions: (1) *agroecosystem services (AES) and disservices (AEDS)*, (2) *social demand*, and (3) *choice experiments*, which set the basis for the theoretical, practical and methodological developments, respectively (Figure 1.1). More specifically, the thesis addresses the non-market value of AES and AEDS, and agricultural practices for their enhancement and mitigation – in the Region of Murcia (south-eastern Spain), estimated through their social demand by using discrete choice experiments. AES and AEDS are socially demanded providing they contribute to human wellbeing.

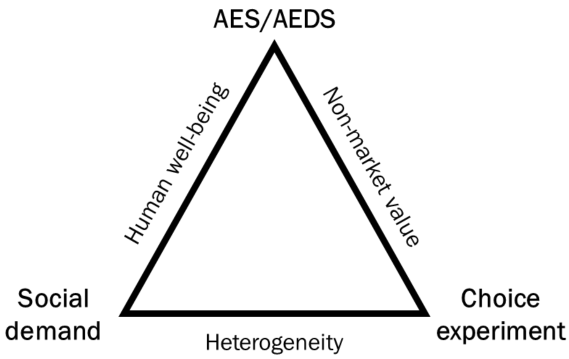


Figure 1.1. Thesis cornerstones

Agroecosystems are anthropised ecosystems for the production of food and fibre, but they also generate a wide set of services and disservices as a result of the ecosystem functioning and human interaction (Fischer and Eastwood, 2016). The wide range of internationally accepted ecosystem service frameworks are mostly centred on natural, non-anthropised, ecosystems – but they are not always greatly adapted to the agroecosystems idiosyncrasies, and even less to their economic valuation. This was the premise that set the starting point of the present thesis. A comprehensive approach for AES and AEDS valuation was proposed and validated by agroecosystem stakeholders as a way to overcome this challenge. The results allowed us to determine the most significant AES and AEDS to be valued in semiarid Mediterranean agroecosystems, with a special focus on the agroecosystems located in the Region of Murcia (south-eastern Spain). Social demand for AES and AEDS was then considered focusing on stakeholders. To do so, a discrete choice experiment was implemented and the choices were assessed by using a conditional logit model. Sources of heterogeneity were therefore address as interaction terms by distinguishing among preferences among different groups of stakeholders.

and tourism) and two AEDS (fresh water and water pollution) were considered to be non-market valued, following the significance previously determined by agroecosystem stakeholders. The results showed the non-linearity of preferences regarding most of these AES and AEDS, as well as the existence of heterogeneity among the respondents, whose target population was the households living in the case study area, the Region of Murcia. A mixed logit model with normal random parameters was employed to tackle social heterogeneity.

Social demand is significant not only to understand the non-market value of AES and AEDS – but also to address the agricultural practices and measures that can shape their provision by agroecosystems. Hence, focusing only on one of the AEDS – water pollution – social preferences for agricultural measures aiming to mitigate such disservice were also estimated. Their social demand is significant because these agricultural practices can modify the provision of other AES and AEDS, thus impacting the overall human wellbeing provided by agroecosystems. Using as a case study the agricultural measures to be implemented in the Campo de Cartagena catchment area to reduce nitrate diffuse pollution from agriculture and their impact on the water quality of the Mar Menor coastal lagoon (Region of Murcia – south-eastern Spain), a discrete choice experiment was implemented to estimate the social demand and non-market value for such measures. Preference heterogeneity was assessed on this occasion by using a latent class mixed logit model, which considered the observed and unobserved heterogeneity.

## 1.2. Background and main challenges

Since the Millennium Ecosystem Assessment report in 2005, the ecosystem service approach has been widely applied to evaluate the contributions that ecosystems provide to people. The concept of ecosystem services, firstly defined as *the benefits people obtain from ecosystems* (MEA, 2005), has been continuously adapted over more than 15 years (Costanza et al., 2017) to encompass the *useful things ecosystems do for people, directly and indirectly,* in a specific context (TEEB, 2010), or, more briefly, the different *contributions that ecosystems make to human wellbeing* (Haines-Young and Potschin, 2018). Ecosystem services comprehend, therefore, all the contributions various ecosystems make to human wellbeing. They comprehend the material and energy resources (provisioning services), the regulation and maintenance of the environmental processes that support human life (regulating services), and the non-material contributions that directly and indirectly impact human wellbeing (cultural services) (Haines-Young and Potschin, 2018). The ecosystem service framework thus reveals the importance of ecosystems and their contributions to human wellbeing.

However, despite the huge development of the ecosystem service framework and the solid theoretical basis supported by the increasing literature and institutional initiatives, such as MEA (2005), TEEB (2010), FEGS-CS (Landers and Nahlik, 2013), IPBES (Pascual et al., 2017), and CICES (Haines-Young and Potschin, 2018), more efforts are needed to adapt the common frameworks to the idiosyncrasies of each type of ecosystem (Costanza et al., 2017; Sandhu et al., 2019). The different ecosystem service frameworks put nature at the heart of the ecosystem functioning, assuming, in most cases, that no human interactions occur in the process of ecosystem services provision. However, many ecosystems have been deeply transformed by humans in such a way that their functioning has, in many cases, totally changed (Palomo et al., 2016). As ecosystem functions are influenced by human activities, the provision of ecosystem services is ultimately affected (Barot et al., 2017).

Agroecosystems are human-based ecosystems whose main purpose is to produce food, fibre, fuel and other material products for consumption and resourcing. Agriculture represents the core activity developed within agroecosystems, making sense of their existence. This involves such a degree of anthropisation that human activities, mainly through agricultural practices, affect the innate functioning of these ecosystems. AES are therefore not fully produced by agroecosystem functioning, and their provision is determined by the level of human activity within each agroecosystem (Mach et al., 2015). Although provisioning AES becomes, *a priori*, the most significant category of AES, regulating and cultural AES are also co-produced by both the natural ecosystem and the human hand (Fischer and Eastwood, 2016). In addition, not all the provided wellbeing contributions by agroecosystems are always positive. Human pressure and its interference in agroecosystem functioning can also lead to negative contributions. First, agricultural practices may impact the current state of agroecosystems, negatively affecting their capacity to provide some AES. Second, they can also lead to the provision of AEDS, which are defined as the "generated functions, processes and attributes that result in perceived or actual negative impacts on human wellbeing" (Shackleton et al., 2016), revealing that agroecosystem contributions can also be harmful. Moreover, interrelationships between AES and AEDS can add complexity to the assessment of agroecosystems given their expected trade-offs (Tancoigne et al., 2014).

In such a context, it is key to ensure that AEDS are included in agroecosystem assessments at the same level as it is done by AES. Integrated assessment of AES and AEDS is needed for at least three reasons. Firstly, considering only AES implies contemplating just part of the overall contribution of the agroecosystem to wellbeing (Schaubroeck, 2017). Secondly, the global assessment of AES and AEDS allows integration of the trade-offs between them and considers the net impact of agroecosystems on human wellbeing (Barot et al., 2017; Blanco et al., 2019). Finally, it helps to achieve a better design of policies intended to produce sustainable and resilient agroecosystems (Sandhu et al., 2019). This becomes more significant when economic valuation is included in the assessment framework. If AEDS are ignored in policy design, this could lead to overestimation of the benefits provided by agroecosystems – which will be translated into suboptimal solutions – and could lead policy makers to make wrong decisions since they have not considered the implied costs.

Economic valuation of AES and AEDS serves to raise awareness of the overall importance of agroecosystems to society and policy makers (De Groot et al., 2012). The economic value of AES and AEDS is merely the translation of the impacts of agroecosystems on human wellbeing into monetary terms. It, therefore, gives researchers and policy makers a way to integrate all agroecosystem contributions into common units to ultimately aid in making substantiated decisions. Both AES and AEDS are valued as long as they provide benefits and costs, respectively, to socioeconomic systems. Benefits and costs may be economic, environmental, or social, and are derived from the direct and indirect use of AES and AEDS, from the option of using them in the future (option value) or even from the mere knowledge of their existence (non-use value) (Pearce and Turner, 1990). The valuation of some AES, such as food provision, is straightforward due to the existence of markets. However, most AES and AEDS are non-marketed and require alternative methods to estimate their values. Stated preference methods, such as contingent valuation and discrete choice experiments, are the preferred methods for this purpose since they allow estimation of the social demand for their provision, and thus the willingness to pay for them, and the value of AES and AEDS variations (TEEB, 2018).

As far as we know, only a few studies have addressed the economic valuation of AES and AEDS from an integrated perspective. Chang et al. (2011) estimated the net value of AES provided by greenhouse vegetable cultivation compared to conventional cultivation, in China. They employed food production, $CO_2$ fixation, soil retention and soil fertility as AES, and irrigation water use, $NO_3^-$ accumulation and $N_2O$ emissions as AEDS. Similarly, Hardaker et al. (2020) estimated the value of agricultural uplands in Wales. They took livestock and crop production, water supply, carbon sequestration and employment as AES flows, and water quality reduction and greenhouse gases emissions as AEDS flows. Sandhu et al. (2020), for their part, estimated the economic value associated with the AES and AEDS provided by corn production systems in Minnesota (US) by adapting the TEEBagrifood framework (TEEB, 2018). Nevertheless, all these authors used direct market and cost-based methods to estimate the economic values of AES and AEDS.

The agroecosystem contributions to human wellbeing can be adjusted by modifying the actual provision levels of AES and AEDS, mainly through agricultural practices. Agricultural practices impact agroecosystem functioning through the pressures they apply, therefore enhancing or reducing the provision of AES and AEDS. This will then be necessarily translated into terms of human wellbeing. For instance, better irrigation practices may reduce the water use in irrigation, while crop diversification or cover crops may increase carbon sequestration and biodiversity, which is expected to provide an increase in utility (Alcon et al., 2020). Therefore, the implementation of agricultural practices may impact wellbeing through the resulting changes in AES and AEDS, in addition to other expected impacts on surrounding ecosystems.

The range of practices that can be applied to enhance the provision of AES or mitigate the production of AEDS is quite wide. In addition, the implementation of agricultural practices is not exempt from increasing or generating new trade-offs among AES and AES and AEDS. A recurrent agricultural practice that illustrates this dilemma is crop fertilization. This agricultural practice is widely implemented by farmers to increase soil fertility, which, in the end, is expected to increase food provision. However, an excess of crop fertilization may result in nutrient pollution of surrounding water bodies and ecosystems, thereby providing AEDS. To overcome this negative outcome, a wide range of alternatives are available for farmers to implement. For instance, eliminating crop fertilization or even reducing farmland. However, this is likely to drastically reduce food provision, namely, the provision of AES. A better alternative, for instance, may be for farmers to reduce nutrient contents in wastewater coming from agroecosystems by using denitrification plants, or by implementing perimeter hedgerows around farms. For their part, these practices are likely to increase the amount of fresh water available for irrigation or environmental purposes, or biodiversity and aesthetic landscape, respectively. This illustrates that challenges regarding the assessment of agroecosystems may arise not only when the focus is on the actual AES and AEDS, but also when new measures – *agricultural practices*- are sought to be implemented.

The selection and implementation of agricultural measures will require tools to assess their impacts. It is easy to find works in the literature that address farmers' willingness to adopt agricultural practices. These are mainly in the context of agri-environmental schemes (Villanueva et al., 2017; Latacz-Lohmann and Breustedt, 2019) and assessments of the cost for farmers to implement such measures (Alcon et al., 2021). However, although the benefits and costs of agricultural practices may imply the entire society, to the best of our knowledge, no work has analysed them from another viewpoint different from the supply-side. It seems that the assessment of social demand for agricultural measures has been disregarded in the literature, although it could be a significant driver for their implementation and success and that these measures may imply social costs and benefits and public expenditure (Smith et al., 2017).

Social demand again plays a crucial role in the selection of agricultural practices that consider social preferences. Public criteria emerge as a supporting tool for decision- makers in their commitment to selecting and implementing a set of agricultural measures to be adopted by farmers. Such public participation becomes even more needed when the perceived impacts derived from the practices are expected to influence surrounding ecosystems under the public domain and with uncertainty in their results. This is the case of implementing agricultural practices to mitigate nutrient pollution. These practices, applied in agroecosystems, are expected to improve water quality in surrounding ecosystems. However, the final effect on water quality might be uncertain. Therefore, the evaluation of costs and benefits considers both the costs for farmers for adopting the agricultural practices and the benefits that society may obtain from both the reaching of a good ecological status on surrounding ecosystems and the implementation of the practices themselves. In addition, since they may involve public investments, the preferences of both farmers and society as a whole for the different measures should be evaluated through cost-benefit analysis before their use in policy-making, guaranteeing the social acceptability of public expenditure.

Preference heterogeneity assessment is key to the public involvement in and the success of new agricultural practices to be implemented. Understanding the factors that motivate the social demand for these types of practices allows policymakers to design agricultural policies which anticipate social support (Ren et al., 2020). Knowledge and understanding of the drivers of social support for new agricultural practices allow better design of socially accepted policies. They allow us to tackle the factors that determine the social support for this kind of policy, and provide information on how to improve policy design and implementation to ensure acceptance by the local population (Fernandes et al., 2019). Policymaking can then focus mainly on these drivers, providing more accurate and reliable values and, in the end, improving the acceptability of agricultural practices' adoption.

The main challenges identified regarding the AES and AEDS valuation can be summarised as follows:

(1) Adaptation of the main ecosystem service frameworks to the particular case of agroecosystems

(2) Identification of the main AES and AEDS to be economically valued in semiarid Mediterranean agroecosystems

(3) Integrated non-market valuation of AES and AEDS considering their social demand

(4) Demand-based valuations of agricultural practices to foster or mitigate AES and AEDS, respectively

(5) Preference heterogeneity regarding social demand for agricultural practices

## 1.3. Objectives and research questions

The current challenges in the AES and AEDS valuation highlight the need to address them in an integrated way, considering their social demand, and following a common approach that considers the idiosyncrasies of agroecosystems. By using this premise, this thesis aims to value economically the integrated social demand of AES and AEDS and the agricultural practices that promote them by adopting a comprehensive approach for agroecosystem valuation in the semiarid Mediterranean region. Hence, this thesis addresses the following research questions:

*Q1. (1) How can we comprehensively consider AES and AEDS? (2) What are the main AES and AEDS in semiarid Mediterranean agroecosystems? What is their relative importance?*

*Q2. (1) What is the non-market value of each of the main AES and AEDS provided by agroecosystems in a semiarid Mediterranean region? (2) What is the total economic value (TEV) of agroecosystems in this area?*

*Q3. (1) Are all the agricultural practices to mitigate nutrient pollution from agriculture equally preferred by society? Is there preference heterogeneity regarding the social demand of agricultural practices? (2) What is the non-market value of each agricultural practice? What is the non-market value derived from the benefits of improving water quality in surrounding ecosystems?*

Theses research questions are tested by (1) modelling the underlying latent utility functions for AES and AEDS (and agricultural practices) derived from the implementation of respective discrete choice experiments, and (2) estimating the consequent willingness to pay for AES and AEDS (and agricultural practices). The discrete choice experiments are applied to the semiarid Mediterranean region, using as a specific case study the Region of Murcia (south-eastern Spain).

## 1.4. Conceptual and methodological framework: An overview

### 1.4.1. Case study: The Region of Murcia in the semiarid Mediterranean region

To address the thesis objectives, the Region of Murcia (south-eastern Spain) is used as a case study. Figure 1.2 shows a map of the case study area. The Region of Murcia, within the Segura River Basin and bordering the Mediterranean coast, is characterised by a semiarid climate with low rainfall (< 400 mm/year) and high mean annual temperatures (between 10 and 18°C), and where long periods of drought are frequent. Hence, water scarcity is also one of its main characteristics. The existence of good-quality soils has fostered the development of a very important agricultural sector here. Indeed, agriculture represents a relevant socioeconomic activity that accounts for more than 5% of the regional GDP (INE, 2020) and nearly 12% of the regional employment (INE, 2021). However, this area is not exempt from environmental challenges, such as water scarcity, groundwater overexploitation, salinisation, and biodiversity loss (Perni and Martínez-Paz, 2017; Esteve-Selma et al., 2016). In sum, these environmental characteristics make the Region of Murcia a case study representative of most semiarid Mediterranean regions (Martínez-Paz et al., 2018).

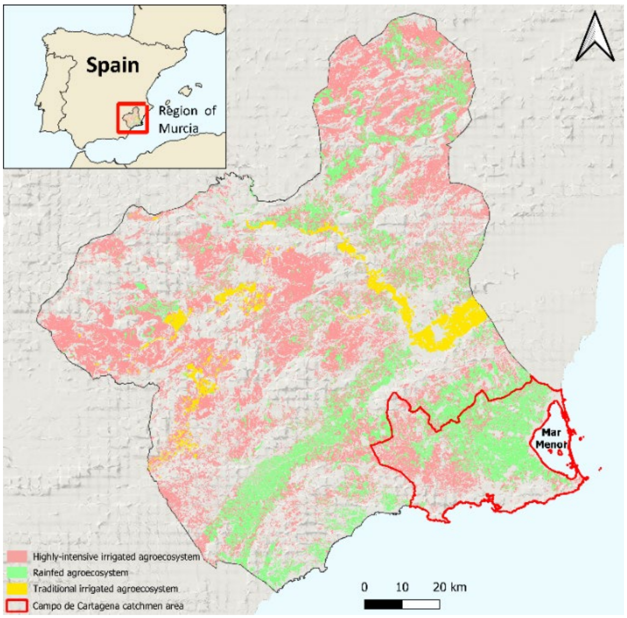


Figure 1.2. Case study area. Region of Murcia (south-eastern Spain)

The agroecosystems within the Region of Murcia can be classified into three different sub-systems regarding their geomorphological and climatic characteristics, water availability and inputs-outputs relations with other ecosystems. There is a rainfed agroecosystem and an irrigated one, which can be further divided into a traditional irrigated agroecosystem (Heider et al., 2018) and a highly-intensive irrigated agroecosystem (Alcon et al., 2017). The rainfed agroecosystem covers around 253,000 ha (CARM, 2019), which represents 57% of the total cropland. Water scarcity determines the crop typology: almonds and olive orchards, as woody crops. Among the herbaceous crops, cereals predominate in the rainfed agroecosystem. Irrigated agroecosystems - traditional (25%) and highly-intensive (75%) - cover 188,000 ha (CARM, 2019). The traditional irrigated agroecosystem follows the Segura River valley, with citrus orchards being the main crop. It is recognisable by its landscape, well-known as the Huerta of Murcia, with high social and cultural values (Martínez-Paz et al., 2019). The highly-intensive irrigated agroecosystem occupies the lowlands, spreading from the south to the north of the region along the Mediterranean coastline. Horticultural crops and citrus are the main crops and their production is export-oriented. Table 1.1 shows the distribution of main crops among agroecosystems in the Region of Murcia (CARM, 2019).

Table 1.1. Crop distribution in agroecosystems of the Region of Murcia

| | Rainfed agroecosystem | | Traditional agroecosystem | | Highly-intensive agroecosystem | | Region of Murcia | |
|---|---|---|---|---|---|---|---|---|
| | ha | % | ha | % | ha | % | ha | % |
| Woody crops | 103,915 | 41.03 | 33,684 | 70.06 | 56,609 | 40.51 | 194,208 | 44.03 |
| Almond | 69,463 | 27.43 | 1,028 | 2.14 | 5,872 | 4.20 | 76,363 | 17.31 |
| Citrus | 0 | 0.00 | 17,692 | 36.80 | 20,890 | 14.95 | 38,582 | 8.75 |
| Herbaceous crops | 49,808 | 19.67 | 4,906 | 10.20 | 56,717 | 40.58 | 111,431 | 25.26 |
| Cereals | 46,533 | 18.37 | 599 | 1.25 | 3,986 | 2.85 | 51,118 | 11.59 |
| Horticultural crops | 0 | 0.00 | 3,747 | 7.79 | 50,942 | 36.45 | 54,689 | 12.40 |
| Fallow land | 99,546 | 39.30 | 9,487 | 19.73 | 26,431 | 18.91 | 135,464 | 30.71 |
| Cultivated area | 253,269 | 100.00 | 48,077 | 100.00 | 139,757 | 100.00 | 441,103 | 100.00 |

In particular, within the highly-intensive irrigated agroecosystem, the Campo de Cartagena catchment area should be highlighted for the purpose of this thesis, given its agri-environmental importance and impacts. The Campo de Cartagena catchment includes 169,450 ha of agricultural land. The main irrigated area comprehends the “Campo de Cartagena” Irrigation Community, which integrates intensive, modern and precision agriculture, yielding fruit and vegetables with high added value. This area finally discharges into the Mar Menor, the largest hypersaline coastal lagoon in Europe. The Mar Menor contains unique habitats, and so is protected at the international level: Natura 2000, Ramsar Wetland, Specially Protected Area of Mediterranean Importance, among others (Perni et al., 2011). Its environmental importance makes the Mar Menor a singular ecosystem to be preserved and protected from its main pressures, which include agriculture, tourism, mining and fishing (Velasco et al., 2018). Focusing on the agricultural sector, the lagoon receives the runoff from the Campo de Cartagena basin in several ephemeral watercourses (*ramblas*) which transport nutrient-enriched water and sediments. Indeed, more than 50% of this nitrate discharge comes from agricultural sources, the value being below 30% for phosphates (Alcolea et al., 2019). Furthermore, the groundwater in the catchment area, which also drains to the Mar Menor, is highly saline due to the presence of excess nutrients from agriculture. All these discharges, together with the insufficient wastewater treatment capacity and the massive tourist influx, have resulted in an increase in the nutrient concentration in the lagoon, finally leading to eutrophication and the generation of algal blooms. This situation first peaked in 2016, when the eutrophication and algal blooms processes worsened, changing the colour of the water, increasing its turbidity and reducing considerably the benthic habitats (Pérez-Ruzafa et al., 2019). However, this was not the only fatal episode, and in October 2019 and August 2021 anoxia caused tons of dead fish to appear on the Mar Menor shore (Perni et al., 2021).

### 1.4.2. Conceptual framework: A comprehensive agroecosystem assessment approach

The conceptual framework followed throughout the thesis is an adaptation of the general approach for ecosystem assessment proposed by Barot et al. (2017). It is based on the “Capacity, Flow, Demand and Pressure” framework (Villamagma et al., 2013) and the TEEB valuation framework, for the particular case of agroecosystems[1]. Figure 1.3 shows a graphical representation of the framework. This conceptual framework assumes that AES and AEDS represent the *flows* from the agroecosystem to the socioeconomic system. AES and AEDS are therefore the links between the biophysical system, the agroecosystem, and the socioeconomic system, where human beings are embedded. The AES and AEDS flows are determined by the *capacity* of the agroecosystem, which is indeed influenced by its functioning and state. When the AES and AEDS flows are provided to the socioeconomic system, they are perceived as benefits and costs, respectively, given their contributions to human wellbeing. Provisioning, regulating and cultural AES and AEDS are in general, but not always, related to economic, environmental and social benefits and costs. Therefore, these benefits and costs are the counterparts of the value people attach for the use, option or non-use of the AES and AEDS flows provided. Human wellbeing is at the core of the socioeconomic system and thereby the main determinant of the *demand* that the socioeconomic system performs. In addition, the socioeconomic system not only can impact the agroecosystem by means of the demand of AES and AEDS it performs but also through the *pressures* that it might put over. The pressures applied to the agroecosystem impact its functioning and health state, affecting, therefore, the agroecosystem capacity to provide AES and AEDS.

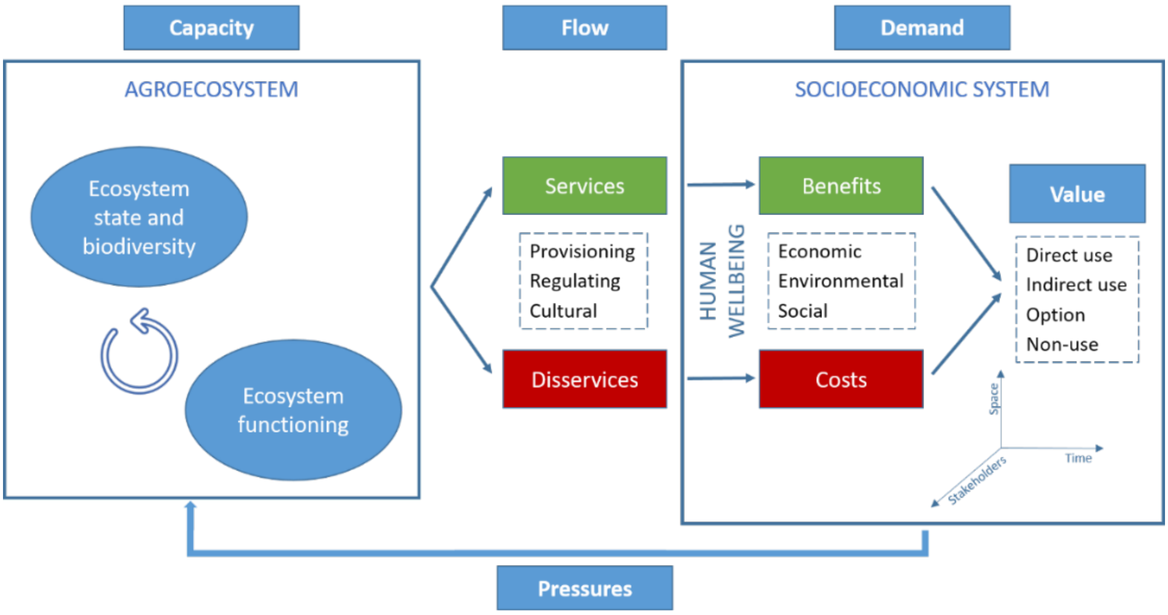


Figure 1.3. Conceptual framework

As illustrated in Figure 1.3, this conceptual framework becomes a continuous cycle, where pressures are applied from the socioeconomic system to modify the agroecosystem

[1] For an in-depth overview of the conceptual framework followed, see *Section 2.2. A comprehensive agroecosystem assessment approach*

functioning, and capacity, and, therefore, the AES and AEDS flows that finally return to the socioeconomic system. Pressures are developed by agricultural stakeholders – mainly farmers, agricultural policy-makers and managers. Agricultural practices are the main means of putting pressure on agroecosystems. Hence, if properly managed, this framework shows a virtuous cycle where agricultural practices play a crucial role in amending the negative, or undesired, outputs from the agroecosystems.

By using a utilitarian approach of human wellbeing based on the random utility theory (McFadden, 1974), the social demand for AES and AEDS is estimated and converted into economic values through their willingness to pay. In addition to the demand of AES and AEDS, agricultural practices might provide desired and undesired effects on the effects on agroecosystem capacity, as well as on the socioeconomic systems, generating trade-offs or increasing the existing ones, among AES and AEDS. In light of this, it not only becomes significant to estimate the value of AES and AEDS, but also the social demand and the non-market value of agricultural practices before being implemented. Agricultural practices - understood as pressures from the socioeconomic system - are defined and implemented by policy-makers and adopted by farmers, although their results, either positive or negative, are perceived by society overall. Hence, the inclusion of public participation is encouraged in the process of selecting those practices to be implemented, those that better suit social preferences.

To illustrate how the framework works, for instance, food is an example of a provisioning AES flow that contributes positively to human wellbeing for the economic and social benefits it provides, whose value is derived from its direct use and option for being used. These values, albeit changeable over time and space, are responsible for the demand put over this AES, and, in the end, over the agroecosystem. Given that this AES provides socially valued benefits, agricultural practices, such as nitrate fertilization are applied to the agroecosystem in order to enhance its provision. This transforms the agroecosystem functioning in such a way that the amount of food provided can increase. However, nitrate fertilization might also be responsible for providing negative outcomes, namely, AEDS. These include nutrient pollution, which is finally converted into poor water quality in surrounding ecosystems. It is perceived in the socioeconomic system as a cost. Therefore, new agricultural practices are socially demanded to mitigate AEDS, applying again pressure over the agroecosystem to modify the provision of AES and AEDS to the socioeconomic system. As stated, the framework for agroecosystem assessment is a cycle where agricultural practices play a key role into improving the contributions of agroecosystems to human wellbeing.

### 1.4.3. Methodological framework: Non-market valuation and discrete choice experiment

#### *1.4.3.1. Non-market valuation*

Agriculture provides more than food. This simple and direct statement hides many challenges for the economic valuation of the agricultural contributions to human wellbeing. As previously stated in the introduction of this thesis, agroecosystems contribute to human wellbeing through the AES and AEDS they provide to the socioeconomic systems, which are valued given the benefits and costs they imply. For instance, in addition to providing food (provisioning AES), agroecosystems contribute positively to human wellbeing through the local climate regulation (regulating AES) or providing an enjoyable landscape for leisure and recreation activities (cultural AES). The negative contributions of agroecosystems encompass, for instance, the pressure on water resources for alternative uses, such as increasing environmental flows of rivers (provisioning AEDS), or the presence of nutrient pollution from agriculture (regulating AEDS). Most of these benefits and costs have some common characteristics that make their economic valuation challenging. Their nature of either public goods, non-rival and non-excludible, or externalities mean that there are no direct markets where, as a result of their trade, a price is obtained. Hence, non-market valuation techniques emerge as a way of assigning monetary value to such benefits and costs.

Environmental economics seeks to adapt the tools and methods of classical microeconomic analysis to contexts where the presence of market failures (public goods, externalities, indefinite property rights and so on) prevents an efficient allocation of resources. Specifically, non-market valuation establishes a monetary value for benefits (costs) that reflects the gain (loss) of human wellbeing due to the increase (reduction) in the provision of AES, or, the reduction (increase) in the provision of AEDS. The economic value of AES and AEDS obtained by using such these non-market valuation techniques is not intended to reflect a real price, but rather a monetary indicator of their wellbeing contribution. In other words, it is an indicator of their relative importance, namely, of how much we are willing to pay for reaching a specific provision level of such AES or for reducing the provision level of such AEDS.

The basics of non-market valuation are grounded in microeconomic theory, which assumes that individuals derive utility from consuming environmental goods and services, such as AES and AEDS. Individuals maximise utility subject to a budget constraint. Hence, the outcome of this optimisation is a set of Marshallian demand functions, which depends on income, prices and non-market environmental outcomes (Haab and McConnell, 2002). Defining an individual's direct utility function, $u$, and indirect utility function, $v$, in terms of a vector of market goods and services, $z$, a vector of AES and AEDS, $q$, and a vector of prices of market goods and services, $p$, the individual chooses the quantity of $z$ that maximises utility subject to income, $y$, and where $q$ is determined exogenously:

$$v(p,q,y) = \max_z\{u(z,q) \mid pz \leq y\} \tag{1.1}$$

Alternatively, the expenditure function associated with the utility change can be used to define the minimum amount of money that an individual spends to reach a desired level of utility:

$$e(p,q,u) = \min_z\{pz \mid u(z,q) \geq u\} \tag{1.2}$$

According to the above optimisations, an individual's utility may be affected by changes in the quantities and qualities of non-marketed AES and AEDS. Two Hicksian welfare measures, the compensating surplus (CS) and the equivalent surplus (ES), are used to measure the wellbeing impact of such changes (Freeman et al., 2014). Hence, if $q$ changes, individual's utility is expected to vary. The value of wellbeing gains due to a change in the quality and/or quantity of the provision of AES and/or AEDS from an initial state $q^0$ (*status quo*) to an improved state $q^1$ is summarised in monetary terms as the CS

$$v(p,q^1,y-CS) = v(p,q^0,y) = v^0 \tag{1.3}$$

And the ES

$$v(p,q^1,y) = v(p,q^0,y+ES) = v^1 \tag{1.4}$$

CS and ES differ in the individual's implied rights to the initial or improved state, respectively. Hence, for instance, the CS implies that the individual has the right to the *status quo*, and therefore the wellbeing gain is valued by maintaining the utility level at the initial state, $v^0$. In consequence, an improvement in the provision of AES, or reduction of AEDS, is measured by the CS as the amount of money that an individual is willing to pay (WTP) as maximum to obtain such wellbeing gain. In the case of the ES, an improvement in $q$ is measured as the minimum amount of money that an individual is willing to accept (WTA) as compensation for not obtaining the improvement. In short, WTP and WTA are theoretically equivalent for a given wellbeing change, that is, the amount of money that makes a person indifferent to an exogenous change in the provision of AES and/or AEDS.

$$WTP = e(p,q^0,u^0) - e(p,q^1,u^0), being\ u^0 = v(p,q^0,y) \tag{1.5}$$

$$WTA = e(p,q^0,u^1) - e(p,q^1,u^1), being\ u^1 = v(p,q^1,y) \tag{1.6}$$

Similarly, both measures can also be applied to a context of wellbeing loss, i.e. due to a reduction in the provision of AES or an increase in AEDS, as Table 1.2 shows (Bateman and Turner, 1993).

Table 1.2. Hicksian measures and WTP/WTA

| | Wellbeing gain | Wellbeing loss |
|---|---|---|
| Compensating surplus (CS) | WTP for environmental improvement | WTA for environmental deterioration |
| Equivalent surplus (ES) | WTA for renouncing to the environmental improvement | WTP for avoiding the environmental deterioration |

The economic valuation of AES and AEDS is commonly performed by using preference-based methods, which rely on human behaviour assuming that values emerge from subjective individual preferences. This approach assumes that the value of AES and AEDS is measured in monetary terms, as a measurement that stablishes the trade-offs among the individual wellbeing perceived by AES and AEDS at the same level as the wellbeing perceived by financial resources. Preference-based methods are comprehended by the total economic value (TEV) framework, which assumes that the value of a given AES and AEDS, and in line, the value of an agroecosystem, is summarised by the sum of the different types of values that compose it. For the purpose of the present thesis, the typology of values can distinguish between use and non-use value, and dividing the former one into three sub-types, direct and indirect use values and option value (Pearce and Turner, 1990). Table 1.3 shows the typology of values according to the TEV framework.

Table 1.3. Typology of values following TEV framework

| Type of value | Sub-type of value | Definition | AES/AEDS |
|---|---|---|---|
| Use | Direct use | Derived from the consumptive or non-consumptive direct use of AES and AEDS | Provisioning and cultural |
| | Indirect use | Derived from its importance for being used for provisioning other AES and/or AEDS | Regulating |
| | Option | Derived from the potential use of AES and AEDS, given that it is expected to be used in the future | Provisioning, regulating and cultural |
| Non-use | | Derived from the current existence of AES and AEDS and/or the possibility that other people from the present or future generation can access the same provision level of AES and AEDS | Cultural |

By using the TEV framework, values for AES and AEDS are elicited from information provided by actual or hypothetical markets where AES and AEDS are traded directly, indirectly or in a simulated way. These possible situations correspond to the three main approaches for valuing AES and AEDS: (1) direct market valuation approaches, (2) revealed preference approaches and (3) stated preference approaches (TEEB, 2010). Table 1.4 shows a brief description of each approach and method, and when to use them.

Among all the valuation methods shown, stated preference methods become the main focus in the thesis. These methods allow us to transform the social preferences for AES and AEDS into monetary terms, providing an economic value for this AES or AEDS. Both these methods - contingent valuation and choice experiment – are based on a market simulation where the AES and AEDS are hypothetically traded. Surveys are employed to simulate such these hypothetical markets, where enumerators represent the supply of AES and AEDS and respondents, the demand. The value of AES and AEDS is therefore the price at which they are traded in this hypothetical market, that is, the WTP to support a certain provision level of AES or to mitigate the provision of AEDS.

Table 1.4. Methods for valuing AES and AEDS under the TEV framework

| Approach | Definition | Method | When to apply | Type of value | AES/AEDS |
|---|---|---|---|---|---|
| Direct market valuation | Direct valuation by using primary markets | Market-based | Real markets for AES/AEDS exist | Direct and indirect use | Provisioning |
| | | Cost-based | Estimated costs that would have been incurred in case of absence, replacing or restoration of the AES/AEDS | | Regulating |
| | | Production function | AES/AEDS used as inputs for the production of goods and services traded in real markets | Indirect use | Provisioning |
| Revealed preferences | Indirect valuation by using secondary markets | Hedonic pricing | AES/AEDS that impact the value of goods and services traded in real markets | Direct and indirect use | Cultural |
| | | Travel cost | AES/AEDS provided when people visit them, with touristic attractiveness | | |
| Stated preferences | Indirect valuation by using simulated hypothetical markets | Contingent valuation | Valuation of AES/AEDS one by one | Use and non-use | Provisioning, regulating and cultural |
| | | Choice experiment | Integrated valuation of AES/AEDS | | |

Specifically, the present thesis applies discrete choice experiments. This method has been selected given the added benefits it provides in comparison with other methods and approaches in the context of the thesis. These may be grouped into four categories. First, choice experiments allow us to value AES and AEDS in an integrated way. That is, choice experiments serve to value the attributes that characterise a specific good or service one by one and the trade-offs among them. Applied to our situation, AES and AEDS are the attributes that compose agroecosystems. This contrasts with contingent valuation, which only allows one to value a specific AES or AEDS, or all the ones provided by an agroecosystem, but without distinguishing the value attached to each AES or AEDS (TEEB, 2018). Second, choice experiments are deeply related to the notion of AES and AEDS, and what they represent. Under a utilitarian approach of human wellbeing, choice experiments convert choices into utility, based on the assumption that individual choices are led by a maximising utility behaviour. This is directly related to the notion of AES and AEDS, which are indeed the contributions of agroecosystems to human wellbeing. Thus, choice experiments link choice human behaviour with AES and AEDS and human wellbeing, by using a utilitarian approach based on the random utility theory. In the next sub-section, this premise will be developed in-depth. Third, albeit in a hypothetical and stated way, choice experiments allow us to estimate social demand for AES and AEDS directly, without using secondary demands for other goods or services with real markets. The use of surveys where individuals are appealed to in order to directly elicit their true preferences for AES and AEDS makes choice experiments a good method for disentangling social demand for goods and services in the context of non-market valuation. Fourth, and related to the previous reason, the use of surveys as a tool for preference elicitation allows for the incorporation of additional questions about attitudes and perceptions that provide better insight on understating social demand for AES and AEDS.

### *1.4.3.2. Discrete choice experiments*

Discrete choice experiments are based on presenting a respondent with a sequence of choice sets, each one composed of a limited set of alternatives. In turn, each alternative is described by a set of attributes, which vary over the alternatives available in the choice set according to their levels. Discrete choice experiments represent the most common elicitation format of stated choice experiments, in which respondents are asked to select only one alternative for each choice set – their preferred one (Champ et al., 2017). Applying this method to the non-market valuation of AES and AEDS, the alternatives represent different agroecosystems that can be found in the semiarid Mediterranean region, the attributes being the AES and AEDS that define such agroecosystems. Similarly, social demand for agricultural practices to mitigate nitrate pollution is estimated using choice experiments where the alternatives are defined by the pairs of agricultural practices and their expected impact on improving water quality in the surrounding ecosystem, which are actually the attributes.

The microeconomic theoretical basis of discrete choice experiments is the random utility maximisation (RUM) model, which defines the assumptions under which individuals follow a utility-maximising behaviour in a context of discrete choices (Freeman et al., 2014). An individual $i$ facing a discrete choice among $J$ alternatives in each of the $T$ choice situations, or choice sets, obtains a certain level of utility $U_{ijt}$ from a specific alternative $j$ in a certain choice set $t$. Therefore, the alternative $l$ is chosen by individual $i$ in choice set $t$ if and only if $U_{ilt} > U_{ijt}, \forall j \neq l$. The individual's utility cannot be directly observed, but the attributes that define the alternatives and some characteristics of the individual can be. Hence, the utility $U_{ijt}$ can be decomposed by a deterministic, $V_{ijt}$, and a stochastic, $\varepsilon_{ijt}$, part, additively integrated (McFadden, 1974):

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt} = \beta X_{ijt} + \varepsilon_{ijt} \tag{1.7}$$

Where $X_{ijt}$ is a vector containing the $k$ attributes (including a cost attribute) and sociodemographic variables that characterise alternative $j$ in choice set $t$ and individual $i$, $\varepsilon_{ijt}$ represents a random term following a joint density over alternatives and choice sets, and $\beta$ a vector of unknown coefficients to be estimated and representing the marginal utility of the attributes. Hence, the WTP for each $k$ attribute, namely, the non-market value of AES and AEDS, is obtained by the marginal rate of substitution (MRS) between the k attribute and the cost attribute. Accordingly, $WTP_k$ is calculated by taking the negative ratio of the attribute's coefficient, $\beta_k$, and the cost coefficient, $\beta_{cost}$, that is, the marginal utility of attribute $k$ and cost, respectively.

$$WTP_k = MRS_{cost}^{k} = \frac{\beta_k}{\beta_{cost}} \tag{1.8}$$

The implementation and development of discrete choice experiments require following a series of steps, widely standardised across environmental literature (Mariel et al., 2021). Figure 1.4 shows a general flowchart with the main steps that follow the implementation and development of the choice experiments employed for this thesis. Nevertheless, this common framework for choice experiment implementation has been adapted to the needs of each specific research within this thesis.

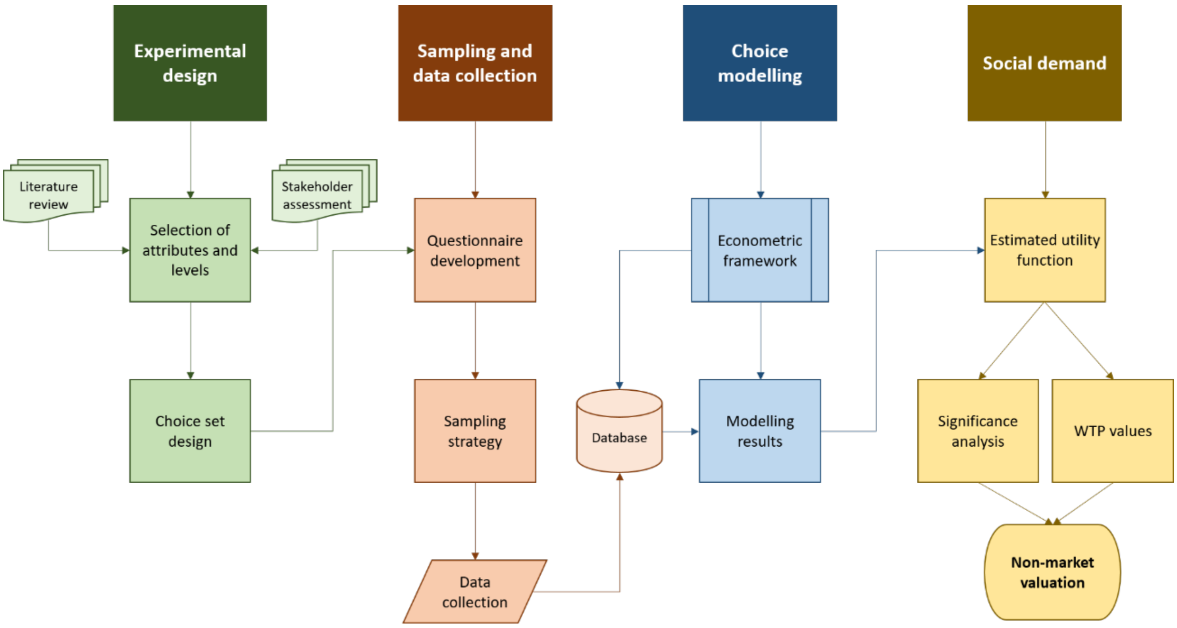


Figure 1.4. Steps followed to implement a discrete choice experiment

The first step in the development of a discrete choice experiment comprises the experimental design. Once the environmental good or service to be valued has been accordingly defined, the attributes and levels that mainly characterise such good or service are selected and described. The selection of attributes is key in this process since the results and conclusions that will be drawn from the choice experiment will be determined by such choices. A literature review and stakeholder assessment (for instance, focus group, direct interviews and meetings) are common tools that usually help with the selection of attributes and their levels. Indeed, in the frame of this thesis, the choice experiment implemented in Chapter 2 is based on an in-deep literature review about ecosystem services and disservices and agroecosystem valuation, while the choice experiment used in Chapter 3 employed the information from the previous choice experiment as a stakeholder assessment.

Attribute levels should be combined in such a way that their variations across choice sets ensure optimality in the relationships among attributes. Respondents should be presented with the trade-offs among attributes that provide the best possible information about their preferences (Hensher et al, 2005). This is the main purpose of choice set design. Orthogonal designs were first to be applied in discrete choice experiments. This type of design is based on the independence of attribute levels, that is, they guarantee no correlation among attribute levels. However, despite orthogonality also ensuring that attribute levels are balanced, these designs might not be optimal when working with non-linear models, such as discrete choice ones. They often include dominated alternatives. Hence, alternative choice set designs were formulated. Efficient designs allow one to obtain more efficient parameter estimates and lower standard errors by optimising a previously defined utility function, which gives more information about the expected choice behaviour. In sum, they seek to minimise the uncertainty, or maximise the informational content, to obtain the most efficient choice set design (Rose and Bliemer, 2009). An *alphabet soup* of efficient designs (Olsen and Meyerhoff, 2017) can be displayed depending on the efficiency measure employed. A-, C-, D-, and S-efficiency criteria can be used to develop the design, depending on the uncertainty measure used to minimise. For instance, the S-efficiency criterion seeks to identify the minimum number of repetitions in the design needed for a parameter to be significant, that is, the minimum sample size to ensure that. All efficiency criteria depend on the parameters of a previously defined utility function. As such, the design will be optimised for these specific parameter values, and therefore, if actual social preferences differ, it cannot be ensured that this will be an optimal design. To overcome this potential issue, some good practices are proposed (Mariel et al., 2021). First, defining prior parameter values based on previous experiences or literature in the area. In addition, performing an initial pre-test experiment based on a non-efficient design is recommended, and using this information as prior values to generate a later efficient design. Second, and complementarily, Bayesian designs are also advised to overcome uncertainty about the true parameter values. This design optimises its results over a larger region of possible parameter estimates, defined according to a prior density instead of a sole parameter value. The use of one or another type of design depends therefore on the quality of the prior information about social preferences. This has been the main determinant of the choice set designs followed in the present thesis. Then, an S-efficient design is presented in the experiments shown in Chapters 2 and 4, while a D-efficient Bayesian design guides the choice experiment performed in Chapter 3.

The second step refers to survey and sampling design and implementation, as well as data collection. One of the main elements of the survey is the questionnaire. A good questionnaire should have a logical order, be easily understandable and, when possible, be quick to be done. It should contain only those questions needed for the research (Dillman et al., 2014). Although there is not a dogma about the structure that a well-designed questionnaire should follow, it is highly recommended that it includes the subsequent items (Mariel et al., 2021), which have been actually followed in the questionnaires for the present thesis: (1) brief introduction of the survey, covering its aim, researchers conducting it, anonymity and treatment of responses, dissemination of results, time taken to answer, among others; (2) behavioural questions related to the AES, AEDS and agricultural practices to be valued; (3) the actual discrete choice experiment, which also includes a detailed description of the context of the valuation, the AES, AEDS and agricultural practices to be valued, respectively, as well as the payment vehicle, and follow-up questions to disentangle strategies in choice patterns, such as protest behaviour; (4) general attitudinal and behavioural questions for explaining preference heterogeneity, such as environmental commitment; (5) sociodemographic information.

The hypothetical nature of the discrete choice experiment method and the employment of surveys for their implementation comprehend many sources of potential biases. Strategic bias, interviewer bias, payment vehicle bias, part-whole bias, hypothetical bias and social desirability bias represent just a few of all the biases that may be entailed by the application of stated preference methods (López-Becerra and Alcon, 2021). Hence, some strategies and approaches are encouraged during the design of the survey and data collection to mitigate them. In particular, within the choice experiments developed in the present thesis, cheap talk was combined with policy consequentiality scripts and budget reminders, when possible, following Penn and Hu's (2019) recommendations for mitigating hypothetical bias. Different trained enumerators were also used as interviewers with the purpose to lessen any presence of interviewer bias. In addition, for the integrated non-market valuation of AES and AEDS in Chapter 3, a forced-choice design with a zero-cost level for the payment vehicle was employed aiming to mitigate both any free-rider behaviour, by systematically choosing a status quo alternative and strategic bias of farmers and agricultural stakeholders by systematically selecting those alternatives with higher attribute levels.

Ensuring the representativeness of the sample and being able to generalise the results often become the main challenges when designing the sampling strategy for discrete choice experiments. Simple random, stratified or probability-based samples are examples of common approaches followed for survey implementation (Mariel et al., 2021). However, they are not the only ones. Indeed, a snowball sampling approach (Biernacki and Waldorf, 1981; Reed et al., 2009) was followed to select relevant stakeholders in the survey implemented in Chapter 2, whilst it was decided to impement a stratified sampling design by county in the choice experiments in Chapter 3 and Chapter 4. Notwithstanding, it is always recommended to check that the final sample is effectively representative of the target population in terms of the main sociodemographic variables.

Data collection is mainly determined by survey mode. Mail, web, telephone and face-to-face surveys, or a mixture of them, represent how choice experiment data can be collected. In principle, there is no rule about what to use; rather, the selected survey mode will depend on the research context (Mariel et al., 2021). In the particular case of this thesis, all the implemented choice experiments were collected by face-to-face interviews using training numerators. This survey mode was selected given its advantages, related to the information transmission, the clarifications to minimise complexity, the capacity of adaptation to respondents and survey duration. The issue that may arise using face-to-face surveys is the interviewer effect, which might bias the results. To overcome this potential issue, the number of interviewers was high and training was provided to them before surveying.

The third main step in the process of developing a choice experiment corresponds to choice modelling. Before obtaining the estimated coefficients that determine the indirect utility function, some basics need to be addressed. The first stage in this process is the codification of attribute levels for modelling (Hensher et al., 2005). Within the present thesis, dummy and continuous coding were employed as strategies for attributes representing categorical and continuous variables, respectively, using a piecewise specification for coding continuous attributes. The codification of attribute levels as continuous allowed us to introduce non-linear terms into the functional form of the utility function. Indeed, this is what is shown in the utility specification of models in Chapter 3, where the use of quadratic and interaction terms between attributes allows us to disentangle diminishing marginal and cross-attribute relationships, respectively. Utility specifications within Chapter 2 and Chapter 4 were assumed linear and additive in attributes, as it is common in the choice experiment literature (Mariel et al, 2021).

Econometric models are used to apply of applying the theoretical RUM models into a pragmatic non-market valuation framework (Train, 2009). The simplest, and most widely used, model is the multinomial logit (MNL) model, alternatively called the conditional logit (CNL) model in practice. It assumes observed homogeneous preferences across individuals. Mixed logit (MXL) and latent class (LC) models, as well as the combination of both (LC-MXL), try to extend the MNL model by allowing for unobserved preference heterogeneity. In short, MXL models assume that heterogeneity follows a continuous or discrete distribution among individuals, whilst LC models group individuals in classes according to their preferences. Other advanced models include the generalised mixed logit (G-MXL) model, which tries to separate preference and scale heterogeneity (Fiebig et al., 2010), and the hybrid choice model, which allows to include latent behavioural variables in the assessment (Ben-Akiva et al., 1999). The maximum simulated likelihood method is the most widely applied for their estimation, although alternatives strategies can also be applied, such as Bayesian analysis or Expectation Maximisation. The selection of the econometric model depends mainly on the purpose of the research and the quality and quantity of the collected data. Hence, as it is stated in the central chapters of the thesis, different econometric models have been employed for the assessment of the choice experiment data. In Chapter 2, the simplest MNL model was selected given the reduced sample size, whilst the MXL model was used in Chapter 3 because of its flexibility to model choice behaviour by considering unobserved heterogeneity. In Chapter 4, the LC-MXL model was selected given the purpose of understanding the sources of preference heterogeneity.

Once the model is estimated, social demand is obtained. This comprises the fourth and last main step in the process of a discrete choice experiment, which includes the interpretation and understanding of the results and the derivation of the WTP values. An overview of how deriving marginal WTP was shown in Equation 1.8. However, often the interest of the research is not only the marginal values of AES and AEDS, or the implementation of just one agricultural practice, but rather the wellbeing implications of policy interventions that involve changes in multiple attributes. Therefore, WTP is not the only deriving value needed, and the assessment should be expanded to include the compensating surplus (CS) in monetary terms. The CS can be derived using the Hanemann utility difference formula (Hanemann, 1984) shown in Equation 1.9.

$$CS = -\frac{1}{\beta_{cost}}\left[ln\left(\sum e^{V_1}\right) - ln\left(\sum e^{V_0}\right)\right] \tag{1.9}$$

Where $\beta_{cost}$ is the marginal utility of income, and $V_0$ and $V_1$ refer to the utility levels before (*status quo*) and after the policy intervention, respectively. Hence, non-market valuation of AES and AEDS, and agricultural practices, can be concluded from a methodological viewpoint.

## 1.5. Thesis outline

This thesis is defended under the category of compendium of publications. It consists of five chapters, among which the three central chapters represent the publications that comprehend such a compendium. The thesis is formed following a logical order in researching, from general to particular. Hence, Chapter 1 embeds the general introduction of the thesis, describing the thesis focus, the research background, the pursued objectives and research questions, and an overview of the conceptual and methodological framework followed in the thesis.

The main body of the thesis is encompassed by the three central chapters that include the publications conforming to the compendium. Chapter 2 corresponds therefore with the first publication and covers the proposal of a comprehensive framework for AES and AEDS valuation as well as their validation and selection of the main AES and AEDS to be valued in semiarid Mediterranean agroecosystems by agricultural stakeholders. A discrete choice experiment was applied to carry out this assessment. Chapter 3 corresponds with the second publication and represents the expected natural progress of Chapter 2 for a non-market valuation of AES and AEDS. This third chapter shows the integrated non-market valuation of AES and AEDS through their social demand by using a choice experiment for the general population. Although it was intended to be primarily practical and oriented to policy recommendations, the results of this chapter allowed us also to establish theoretical implications. The quadratic functional form of the utility function over some AES and AEDS was verified, revealing the presence of diminishing marginal utility and cross effects among the social value of some AES and AEDS. Chapter 4 focuses on social demand for specific agricultural practices to mitigate the provision of AEDS, in particular, nutrient pollution from agriculture. Again, a discrete choice experiment was developed for the population of the Region of Murcia (south-eastern Spain), as a specific case study of the semiarid Mediterranean region. The expected improvement in water quality in a surrounding water ecosystem, the Mar Menor, was jointly valued. In addition to the practical and policy implication of these results, the focus was on the assessment of preference heterogeneity.

Finally, Chapter 5 forms the final part of the thesis and provides a synthesis of the main results, answering the research questions and showing to what extent the objectives have been achieved. In addition, an integrated discussion and policy implications are displayed together with the main remarks of this thesis.

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# Chapter 2. A comprehensive approach for agroecosystem services and disservices valuation

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## Highlights

- Ecosystem services paradigms are adapted to agroecosystem idiosyncrasies.
- A proposed agroecosystem services and disservices valuation approach is validated.
- Relevant agroecosystem services and disservices to be valued are identified.
- Integrated valuations of agroecosystem services and disservices are needed.

## Abstract

The use of the ecosystem services approach for ecosystem management, including the valuation of ecosystem services, has grown in recent decades. Although a common framework is used, each ecosystem has its own characteristics. The agroecosystem, for example, is an anthropised ecosystem where ecosystem service flows are highly interrelated with the environment, positively or negatively. Therefore, agroecosystem services are usually accompanied by disservices. The valuation of agroecosystem services and disservices requires adaptation of existing ecosystem services paradigms to accommodate the innate agroecosystem idiosyncrasies. To this end, in this study, a comprehensive approach for valuation of agroecosystem services and disservices was proposed and validated in a semiarid western Mediterranean agricultural area through stakeholder assessment, using a choice experiment. The results suggest that all categories of services (provisioning, regulating, and cultural) should be taken into account when valuing agroecosystem services and disservices. In particular, food provision (a provisioning service), water (a provisioning disservice), local climate regulation and biodiversity (regulating services), water purification and waste treatment (regulating disservices), and recreation and tourism (cultural services) are relevant for this purpose. Their relative importance in agroecosystems valuation reached 70% for agroecosystem services and 30% for disservices. Specifically, biodiversity (38%) emerged as the most relevant agroecosystem service to be valued, followed by recreation and tourism (20%), local climate regulation (7%), and food provision (5%). Among the agroecosystem disservices, water and waste treatment (15%), and water purification (15%) together contributed to 30% of the total importance. Agroecosystems should be valued considering their multifunctional character and the integration of agroecosystem services and disservices.

Keywords: Anthropised ecosystems; Choice experiment; Mediterranean agroecosystems; Stakeholder assessment; Human wellbeing.

## 2.1. Introduction

The ecosystem services approach highlights the importance of nature's contribution to human life and wellbeing. The notion of ecosystem services reveals that human wellbeing closely depends on the ecosystems in which humans exist. Ecosystem functioning impacts human wellbeing through the ecosystem services provided. Ecosystems may supply food, fuel, or fibre (provisioning services), contribute to the regulation of natural functions (regulating services), or even provide an environment for leisure activities (cultural services). Thus, ecosystem services represent ecosystem flows that are ultimately perceived as contributions to human wellbeing.

Over the past two decades, both development and extension of the ecosystem services approach have been encouraged through growth of the related literature and international institutional support. From "the benefits people obtain from ecosystems" (MEA, 2005) to "the contributions that ecosystems make to human wellbeing" (Haines-Young and Potschin, 2018), the definition of ecosystem services in the scientific literature has been adapted over time to incorporate the advances achieved. Initiatives such as MEA (2005), TEEB (2010), FEGS-CS (Landers and Nahlik, 2013), IPBES (Pascual et al., 2017), and CICES (Haines-Young and Potschin, 2018) reflect this development process. Most of these initiatives have served to establish a solid theoretical basis for the definition and classification of ecosystem services and the impact of ecosystems on human wellbeing (Costanza et al., 2017). However, despite their wide use and extension, these definitions and classifications may not fit all types of ecosystems (Fisher et al., 2009; Ojea et al., 2012). The different ecosystem services frameworks developed assume that ecosystem service flows arise from natural processes and no human interactions are considered within the ecosystem functions. However, many ecosystems have been deeply transformed by humans, in such a way that their functioning has, in many cases, totally changed (Palomo et al., 2016). As ecosystem functions are influenced by human activities, it ultimately affects the provision of ecosystem services (Lele et al., 2013). Therefore, in anthropised ecosystems, such as agroecosystems, the flow of services should be carefully considered (Barot et al., 2017).

Agroecosystems are created by humans to provide a specific provisioning service. This involves such a degree of anthropisation that human activities, mainly through agricultural practices, affect the innate functioning of these ecosystems. Therefore, agroecosystem services are not fully produced by agroecosystem functioning, and their provision is determined by the level of human activity within each agroecosystem (Mach et al., 2015). Agroecosystem services are, therefore, co-produced by both the natural ecosystem and the human hand (Fischer and Eastwood, 2016). In addition, this human interference may not always have the desired positive outcomes (Barot et al., 2017). First, agricultural practices may impact the current state of agroecosystems, negatively affecting their capacity to provide agroecosystem services (AES). Second, they can also lead to the provision of agroecosystem disservices (AEDS), which are defined as the "generated functions, processes and attributes that result in perceived or actual negative impacts on human wellbeing" (Shackleton et al., 2016), revealing that agroecosystem contributions can also be harmful. Furthermore, interrelationships between AES and AEDS are expected in agroecosystems, providing many more trade-offs among them than win-win solutions. In turn, these trade-offs are promoted by human practices, which add complexity to the assessment of agroecosystems (Tancoigne et al., 2014).

The expression of the value of AES and AEDS serves to raise awareness of the overall importance of agroecosystems to society and policy makers (De Groot et al., 2012). Both AES and AEDS are valued as long as they provide benefits and costs, respectively, to socioeconomic systems. Benefits and costs may be economic, environmental, or social, and are derived from the direct and indirect use of AES and AEDS, from the option of using them in the future (option value) or even from the mere knowledge of their existence (non-use value) (Pearce and Turner, 1990). In addition, these values are context-dependent (Díaz et al., 2018). Time, spatial scale, cultural background, and stakeholder involvement are key elements that determine the values of AES and AEDS received by a socioeconomic system. It is well known that agroecosystems provide benefits and costs to society, but there is no consensus in the literature regarding the main AES and AEDS that should be valued. In fact, recent advances in AES valuation refer to specific AES without a common agreement. Rodríguez-Entrena et al. (2014) and Granado-Díaz et al. (2020) focused on the economic valuation of erosion control, carbon sequestration, and biodiversity in olive agroecosystems in Andalusia (southern Spain); Divinsky et al. (2017) valued food provision, pollination, and landscape on an experimental farm in Galilee (NE Israel); and Bernués et al. (2019) assessed the social demand for quality food products, fire control, biodiversity, and landscape in mountain agroecosystems in Huesca (NE Spain).

In this context, this study aimed to identify the AES and AEDS that should be valued, considering the innate idiosyncrasies of agroecosystems. To this end, a comprehensive agroecosystem assessment approach was proposed and validated using a stakeholder choice experiment. The Region of Murcia (south-eastern Spain) was used as a case study because it is representative of semiarid western Mediterranean agroecosystems.

The novelty of this research lies in its integration of anthropisation, AES, and AEDS into a common approach for agroecosystem valuation, while it also considers the overall complex relationships between the biophysical and socioeconomic systems. Therefore, this study aids in filling the knowledge gap regarding the integrated valuation of AES and AEDS, and its implications are expected to be useful for research purposes and decision-making. First, the proposed framework enables the adaptation of the main ecosystem services paradigms to the specifics of agroecosystems. Second, the results of a stakeholder assessment elucidate the relative relevance of the AES and AEDS valued in a semiarid western Mediterranean context. Thus, this study provides researchers with baseline information to value the overall contributions of agroecosystems to human wellbeing. It also provides policy makers with background information that should enable them to focus on the AES and AEDS that need to be better managed.

In the following section of this paper, a comprehensive approach for agroecosystem assessment is proposed. Sections 3 and 4 describe the validation of this approach through a stakeholder choice experiment, including the methodology used and the results, respectively. In Section 5, the results and their implications are discussed and Section 6 is the conclusion of the paper.

## 2.2. A comprehensive agroecosystem assessment approach

In an anthropised ecosystem, the integrated valuation of AES and AEDS requires a framework that considers both positive and negative impacts on human wellbeing. To achieve this, the Barot et al. (2017) framework for anthropised ecosystems has been applied. In addition, the main paradigms for ecosystem services, such as MEA (2005), TEEB (2010), and CICES (Haines-Young and Potschin, 2018), have been revised and readapted to the innate idiosyncrasies of the agroecosystem. Barot et al. (2017) adapted the *Capacity, Flow, Demand, and Pressure* framework developed by Villamagma et al. (2013) to include anthropisation and the presence of disservices within a common framework. This framework assumes that agroecosystem functioning depends on the state of health of the agroecosystem as well as on the biodiversity and the innate processes and functions within the agroecosystem (Figure 2.1). Agroecosystem functioning comprises the agroecosystem functions that form the basis of production of AES and AEDS flows. The joint consideration of agroecosystem state, biodiversity, and functioning determines the potential of an agroecosystem to provide AES and AEDS, namely, the agroecosystem capacity. Agroecosystem functioning impacts human wellbeing by means of the AES and AEDS provided, which are therefore considered the flows from the agroecosystem to the socioeconomic system. However, within the socioeconomic system, AES and AEDS represent benefits and costs, respectively, which could be translated into economic values by means of market and non-market valuation methods.

At this point, it should be stated that the benefits and costs related to AES and AEDS are not fixed over space and time but depend on the context in which they are framed (Díaz et al., 2018). The demand for AES and AEDS, that is, the amount of services and disservices desired, is determined by the entire society, and consequently by sociocultural preferences. Preferences, which are assumed to be invariant in a specific context (Braga and Starmer, 2005), are ascertained by both the concrete agroecosystem and the sociocultural frame within which they are evaluated (Bernués et al., 2019; Alcon et al., 2020). Consequently, stakeholders also play a key role in creating demand within the socioeconomic system. Their actions may have an influence not only on the value that AES and AEDS provide, but also, and mainly, on the way an agroecosystem is managed. This management will in turn affect the functioning of the agroecosystem and thus its capacity to generate AES and AEDS flows. Agricultural practices represent the main human pressure on an agroecosystem.

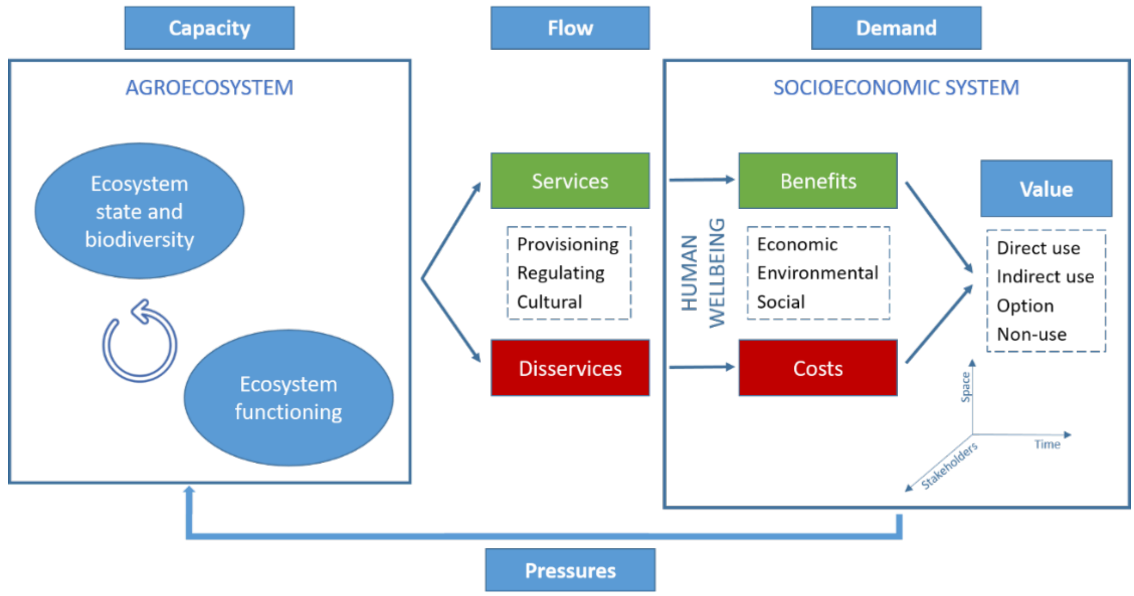


Figure 2.1.Conceptual approach linking agroecosystem functioning, services and disservices, value and agricultural practices. It is based on the "Capacity, Flow, Demand and Pressure" framework (Villamagma et al., 2013) and the TEEB valuation framework.
Source: Own elaboration, adapted from Barot et al. (2017) and TEEB (2010).

The proposed approach seeks to capture the innate idiosyncrasies of agroecosystems. Trade-offs between AES and AEDS are expected within an agroecosystem, and are influenced by pressures (Barot et al., 2017). For instance, food provision, which is the core service provided by agroecosystems, can be enhanced by agricultural practices (pressures), such as fertiliser application, but this may imply the emission of contaminants to the atmosphere and water bodies, thus providing AEDS. Consequently, not only the capacity of the agroecosystem could be affected, but also its functioning, which ultimately affects the current AES and AEDS flows. Furthermore, trade-offs and pressures reduce the capacity of an agroecosystem to provide the maximum level of AES. Human demand also influences the capacity and functioning of an agroecosystem through pressures; therefore, the provision of AES and AEDS is a consequence of the relationship between humans and nature. A final point to note is that this approach focusses only on the agroecosystem, without considering connections between it and surrounding ecosystems beyond the AES and AEDS provided. Consequently, land use changes, which may imply transition from or to other ecosystem types (e.g. from forest to agroecosystems) are not considered.

Our approach encompasses two different but interrelated systems: the biophysical system, which corresponds to the agroecosystem, and the socioeconomic system, centred on human wellbeing (Figure 2.1). In the biophysical system, the proposed approach can also be translated to the existing typologies of ecosystem services. In the socioeconomic system, AES and AEDS are treated as benefits and costs, respectively, according to their impact on human wellbeing. Table 2.1 summarises how the most widely applied ecosystem services paradigms are connected as well as how they were readapted to the specific case of the agroecosystem. For this purpose, a chronological order was followed from MEA (2005) to CICES (Haines-Young and Potschin, 2018).

Regarding provisioning services, the main classifications agree on and recognise different types of ecosystem services, not just food provision. However, when the agroecosystem is considered, food provision could be assessed as one of the AES, and water as one of the AEDS. Of the AES, food is the only one considered in the assessment of agroecosystems because the other ecosystem services all translate into production outputs. In the case of water ecosystem services, however, agroecosystems do not provide fresh water to other ecosystems, but instead utilise water from them, consequently decreasing the availability of water in these other ecosystems (Strzepek and Boehlert, 2010). Water demand for agricultural purposes has become a pressure for freshwater ecosystems. Environmental flows (e-flows) might be therefore undermined by agroecosystem functioning, which in turn could be translated into a depletion of water ecosystem services and may compromise the sustainability and wellbeing of organisms that depend on these ecosystems (Kuriqi et al., 2019, 2020; Zheng et al., 2020). For such reasons, agroecosystems provide water AEDS rather than water AES.

Regulating services represent the widest category of AES. Our approach also includes some supporting services, in the case of MEA (2005), and habitat services, in the case of TEEB (2010); specifically, only those services that would not imply double-accounting bias. Regarding air quality regulation, all the classifications concur that ecosystems contribute positively to improvements in air quality, diminishing the contaminants of human origin. However, this is not the case for agriculture, which may be responsible for emitting contaminants to the atmosphere, such as ammonia or nitrous oxide, derived mainly from fertiliser application (Tubiello et al., 2015). Thus, the contribution of agroecosystems to air quality regulation is expected to be negative and should be considered as one of the AEDS. Climate regulation is broadly recognised as being among the AES in all classifications. The contribution of ecosystems to climate regulation can be considered both globally and locally because ecosystem functioning can impact the global carbon cycle dynamics as well as the thermodynamics and weather in the locations where the ecosystems exist. Therefore, an agroecosystem could contribute to carbon sequestration, both in the soil and by crop photosynthesis, which would form part of the agroecosystem contribution to global climate regulation (González-Sánchez et al., 2012). In addition, an agroecosystem could also contribute to temperature regulation, which is part of local climate regulation (Albaladejo-García et al., 2020).

Although water regulation is included within the main ecosystem services paradigms, it cannot be applied in the case of agroecosystems because the agroecosystem contribution to the regulation of water cycle dynamics is relatively insignificant compared to that of other ecosystems. In addition, water regulation, which includes evapotranspiration, infiltration, and runoff, is closely related to agroecosystem functioning, water supply, and water purification, generating service overlapping and double-counting biases when it is valued (Ojea et al., 2012). However, agroecosystems may interfere with water purification and waste treatment. Agricultural soils provide water purification, preventing the filtering of nutrients to aquatic ecosystems (Schröder et al., 2020). This agroecosystem function could also be enhanced by agricultural practices, such as cover crops (Skaalsveen et al., 2019) or the inclusion of buffer strips to delimit cropland (Terrado et al., 2015), thereby providing AES. In addition, other agricultural practices, especially fertiliser application, which have been considered as pressures in our approach, may be responsible for water pollution. The runoff and leaching of water following excessive use of nitrogen fertiliser generates diffuse pollution from agriculture, which contributes directly to the salinisation of groundwater and may negatively affect other ecosystems (Jiménez-Martínez et al., 2016), thereby providing AEDS.

Ecosystem functioning contributes to soil conservation and quality in different ways, including erosion prevention, soil formation, and soil fertility. The state of the soil is crucial to agroecosystem functioning, and agricultural practices may affect this. Therefore, the agricultural contribution to soil maintenance may be positive or negative depending on the management practices. Intensive tillage, quite common in traditional agriculture, may generate high erosion rates and, therefore, AEDS (Montgomery, 2007). However, more environment-friendly practices such as crop diversification or green manure use can boost soil organic matter, increase fertility, and thus contribute to soil maintenance (Morugán-Coronado et al., 2020), implying the provision of AES. Therefore, agroecosystems can contribute positively and negatively to soil, providing both AES and AEDS, respectively.

Biological control of diseases and pests, and pollination are considered in most ecosystem services classifications. However, the case of the agroecosystem is again quite different because agroecosystems receive biocontrol agents and pollinators from other ecosystems. As Zhang et al. (2007) and Power (2010) suggested that biological control and pollination are ecosystem services provided to an agroecosystem by natural habitats. These external services allow agroecosystems to maintain the provision of AES flows. Nevertheless, agricultural practices can also impact the provision level of biological control and pollination ecosystem services. Conservation agriculture and crop diversification are two examples of agricultural practices that have positive impacts on biological control and pollinators (Aguilera et al., 2020). Conversely, agricultural intensification, mainly through pesticide and fertiliser application, is responsible for the decrease in pollinators worldwide (Potts et al., 2010; Main et al., 2020) as well as for the loss of soil and plant biodiversity (Culman et al., 2010; Beeckman et al., 2018). Based on this, agriculture does affect the maintenance of genetic diversity within an agroecosystem, and therefore agroecosystems contribute to biodiversity (Paiola et al., 2020). Impacts on biodiversity could be both positive and negative depending on the particular agricultural practices. Therefore, agriculture may promote or reduce the biodiversity that develops within an agroecosystem (Martin et al., 2019), providing both AES and AEDS.

Agroecosystems may contribute to the regulation of extreme events, such as floods, by improving the resilience of ecosystems. Resilience, defined as the ability of systems (either ecosystems or socioeconomic systems) to maintain their original functioning and capability after exposure to a disruptive change (Holling, 1973), is key to ensuring the long-term sustainability of agroecosystems themselves, but, above all, of their surrounding ecosystems and socioeconomic systems. The capacity of agroecosystems to moderate extreme events, mainly through the capability of crops and vegetation to retain and store water, may mitigate the consequences of climate change, such as heavy rainfall, floods, and drought. Resilience, in this sense, should be understood not only as the moderation of extreme events, but also as a positive contribution to the human wellbeing derived from it (Qiu, 2019). The focus is on mitigating the negative effects that disruptive changes would produce in the absence of agroecosystems. Therefore, agroecosystems can mitigate the negative impacts of extreme events on surrounding ecosystems and socioeconomic systems, and thereby enhance the resilience of these systems. This enhanced resilience should also be considered as a service provided by agroecosystems (Peterson et al., 2018).

Cultural AES should also be included in the assessment of agroecosystems in order to add the non-material benefits that agroecosystems provide to society (Huber and Finger, 2019). These benefits could be obtained through spiritual and cultural values, aesthetic values, opportunities for recreation, tourism, and cognitive development. They fit completely with the main ecosystem services classifications.

Table 2.1. Main ecosystem service classifications and proposal for agroecosystems.

| | MEA (2005) | TEEB (2010) | CICES (Haines-Young and Potschin, 2018) (Division \| Group) | Agroecosystem proposal (AES/AEDS) |
|---|---|---|---|---|
| Provisioning services | Food | Food | Biomass \| Cultivated terrestrial plants for nutrition, materials or energy production | Food (AES) |
| | Fibre | Raw materials, ornamental resources | | |
| | Biochemicals | Medicinal resources | | |
| | Genetic resources | Genetic resources | Genetic material from all biota (including seed, spore or gamete production) \| Genetic material from plants, algae or fungi | |
| | Fresh water | Water | Water \| Surface/Groundwater used for nutrition, materials or energy production | Water (AEDS) |
| Regulating services | Air quality regulation | Air quality regulation | Transformation of biochemical or physical inputs to ecosystems \| Mediation of nuisances of anthropogenic origin | Emissions of contaminants to the atmosphere (AEDS) |
| | Global climate regulation | Climate regulation | Regulation of physical, chemical and biological conditions \| Atmospheric composition and conditions | Global climate regulation (AES) |
| | Local climate regulation | Climate regulation | Regulation of physical, chemical and biological conditions \| Atmospheric composition and conditions | Local climate regulation (AES) |
| | Water regulation | Regulation of water flows | Regulation of physical, chemical and biological conditions \| Water conditions | (Not included – *service overlapping*) |
| | Water purification and waste treatment | Waste treatment | Transformation of biochemical or physical inputs to ecosystems \| Mediation of wastes or toxic substances of anthropogenic origin by living processes | Water purification and waste treatment (AES/AEDS) |
| | Erosion regulation | Erosion prevention | Regulation of physical, chemical and biological conditions \| Regulation of baseline flows and extreme events | Soil maintenance (AES/AEDS) |
| | Soil formation (supporting) | Maintenance of soil fertility | Regulation of physical, chemical and biological conditions \| Regulation of soil quality | |
| | Disease regulation | Biological control | Regulation of physical, chemical, biological conditions \| Pest and disease control | Biodiversity (AES/AEDS) |
| | Pest regulation | | | |
| | Pollination | Pollination | Regulation of physical, chemical and biological conditions \| Lifecycle maintenance, habitat and gene pool protection | |
| | - | Maintenance of lifecycles of migratory species, maintenance of genetic diversity | Regulation of physical, chemical and biological conditions \| Lifecycle maintenance, habitat and gene pool protection | |
| | Natural hazard regulation | Moderation of extreme events | Regulation of physical, chemical and biological conditions \| Regulation of baseline flows and extreme events | Resilience (AES) |
| Cultural services | Spiritual and religious values | Inspiration for culture, art and design, spiritual experience | Indirect, remote, often-indoor interactions with living systems that do not require a presence in the environmental setting \| Spiritual, symbolic and other interactions with the natural environment | Culture, art and design (AES) |
| | Aesthetic values | Aesthetic information | Direct, *in-situ* and outdoor interactions with living systems that depend on a presence in the environmental setting \| Intellectual and representative interactions with the natural environment | Aesthetic values (AES) |
| | Recreation and ecotourism | Opportunities for recreation and tourism | Direct, *in-situ* and outdoor interactions with living systems that depend on a presence in the environmental setting \| Physical and experiential interactions with the natural environment | Opportunities for recreation and tourism (AES) |
| | | Information for cognitive development | Direct, *in-situ* and outdoor interactions with living systems that depend on a presence in the environmental setting \| Intellectual and representative interactions with the natural environment | Cognitive development and good living (AES) |

The socioeconomic system of the proposed approach (Figure 2.1) focusses on how AES and AEDS are perceived as benefits and costs, respectively. The interrelationships between the biophysical and socioeconomic systems show how provisioning, regulating, and cultural AES and AEDS are perceived as economic, environmental, and social benefits and costs, respectively, in the socioeconomic system, having an economic value. The economic value is derived from their direct and indirect use, the option of their use in the future, and even their mere existence (non-use value) (Pearce and Turner, 1990).

Table 2.2. AES and AEDS, typology of benefits/costs and values.

| | Agroecosystem (AES/AEDS) | Benefit | Cost | Type of benefit/cost: Economic | Environmental | Social | Type of value: Direct use | Indirect use | Option | Non-use |
|---|---|---|---|---|---|---|---|---|---|---|
| Provisioning services | Food (AES) | x | | x | | x | x | | x | |
| | Irrigation water (AEDS) | | x | x | x | | x | | x | |
| Regulating services | Emissions of contaminants to the atmosphere (AEDS) | | x | | x | | | x | x | |
| | Global climate regulation (AES) | x | | | x | | | x | x | |
| | Local climate regulation (AES) | x | | | x | | | x | x | |
| | Water purification and waste treatment (AES/AEDS) | | x | x | x | x | | x | x | x |
| | Soil maintenance (AES/AEDS) | x | x | x | x | | | x | x | |
| | Biodiversity (AES/AEDS) | x | x | | x | x | x | x | x | x |
| | Resilience (AES) | x | | | x | | | x | x | |
| Cultural services | Culture, art and design (AES) | x | | | | x | | x | x | x |
| | Aesthetic values (AES) | x | | | | x | x | | x | x |
| | Opportunities for recreation and tourism (AES) | x | | x | | x | x | | x | |
| | Cognitive development and good living (AES) | x | | x | | x | x | | x | |

Table 2.2 shows the links between the proposed AES and AEDS and their respective type of benefit and cost, and their type of value (TEEB, 2010). Provisioning AES and AEDS are mostly related to economic benefits and costs, while regulating and cultural AES and AEDS are linked to environmental and social benefits and costs, respectively. However, these relationships are not always so straightforward. For instance, provisioning AEDS may also provide environmental costs, while cultural AES may generate economic benefits. Regarding value, provisioning AES and AEDS tend to be valued in relation to their direct use, regulating AES and AEDS are more related to indirect use and option values, and cultural AES and AEDS mostly refer to direct use, option, and non-use values.

## 2.3. Methodology

### 2.3.1. Case study

This case study was located in the Region of Murcia (south-eastern Spain), within the Segura River Basin (Figure 2.2). This region is characterised by a semiarid climate with low rainfall and long periods of drought which generate agri-environmental challenges such as water scarcity, groundwater overexploitation, salinisation, and biodiversity loss. Agriculture represents a relevant socioeconomic activity which accounts for more than 5% of the regional GDP (INE, 2018) and nearly 12% of the regional employment (INE, 2019).

The agroecosystems within the case study area are based on a dual system, where irrigated and rainfed agriculture coexist. The irrigated agroecosystems comprise 188,000 ha (CARM, 2019) of high-productivity fruit and horticultural crops (Alcon et al., 2017), and these, in turn, are divided into two agricultural subsystems: traditional and intensive. The Segura River valley hosts traditional irrigation (Heider et al., 2018), while intensive irrigation occurs further away from the river (Alcon et al., 2021). The rainfed agroecosystem is distinguished by low profitability, with almonds the main crop. It covers approximately 253,000 ha (CARM, 2019), distributed throughout the case study area. It should be noted that some aquifers and a coastal ecosystem, the Mar Menor lagoon, are influenced by agricultural flows in the region. The agri-environmental and socioeconomic characteristics, as well as the blend of different and interdependent agroecosystems, make the Region of Murcia a representative case study for the semiarid western Mediterranean area (Martínez-Paz et al., 2018).

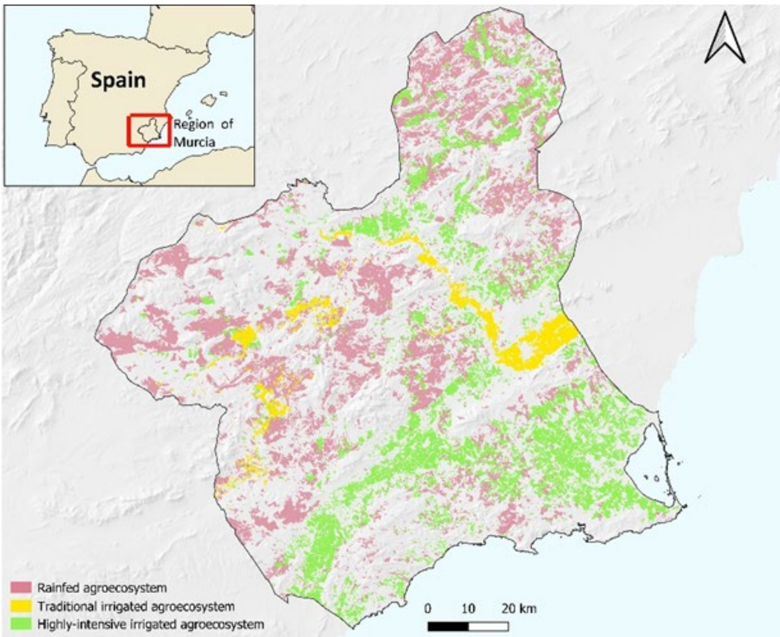


Figure 2.2. Case study. Region of Murcia (south-eastern Spain).

### 2.3.2. Choice experiment method

The choice experiment is a stated preference method based on the multi-attribute utility theory (Lancaster, 1966) and the random utility theory (McFadden, 1974). Accordingly, an agroecosystem can be defined as a set of AES and AEDS, and individuals can choose the preferred agroecosystem alternative according to their expected utility level. Choice experiment applications in agri-environmental valuation tend to include a status quo (SQ) or opt-out alternative that reflects the current situation, where no action is taken (Barreiro-Hurle et al., 2018). In our case study, the SQ alternative was the rainfed agroecosystem, which is less human-managed than the irrigated agroecosystem.

The choice experiment method is appropriate for selecting the most relevant AES and AEDS for valuation, because it allows the modelling of individual discrete choices among different AES and AEDS, and even among different agroecosystems. It is important to note that a wide range of methods could be used to attain the research objective, including multi-criteria analysis and, more specifically, the analytic hierarchy process. However, these methods only allow us to consider all the AES and AEDS trade-offs through pairwise comparisons, and not in an integrated way, despite providing similar results (Kallas et al., 2011). The choice experiment method has been widely applied to AES valuation (Rodríguez-Entrena et al., 2014; Bernués et al., 2019), but has been slightly used for stakeholder assessment (Villanueva et al., 2017).

Developing and implementing a choice experiment involves a five-step process (Hoyos, 2010): (1) selection and definition of attributes and levels, (2) choice set design, (3) questionnaire development, (4) sampling strategy and data collection, and (5) assessment of choices and modelling of results. The first two steps are summarised in Section 2.3.2.1 (Experimental design). Questionnaire development is covered in Section 2.3.2.2 (Sampling and data collection). Section 2.3.2.3 describes how the assessment of choices and modelling of results was accomplished in relation to an econometric framework. Figure 2.3 shows a flowchart of the choice experiment development and implementation process.

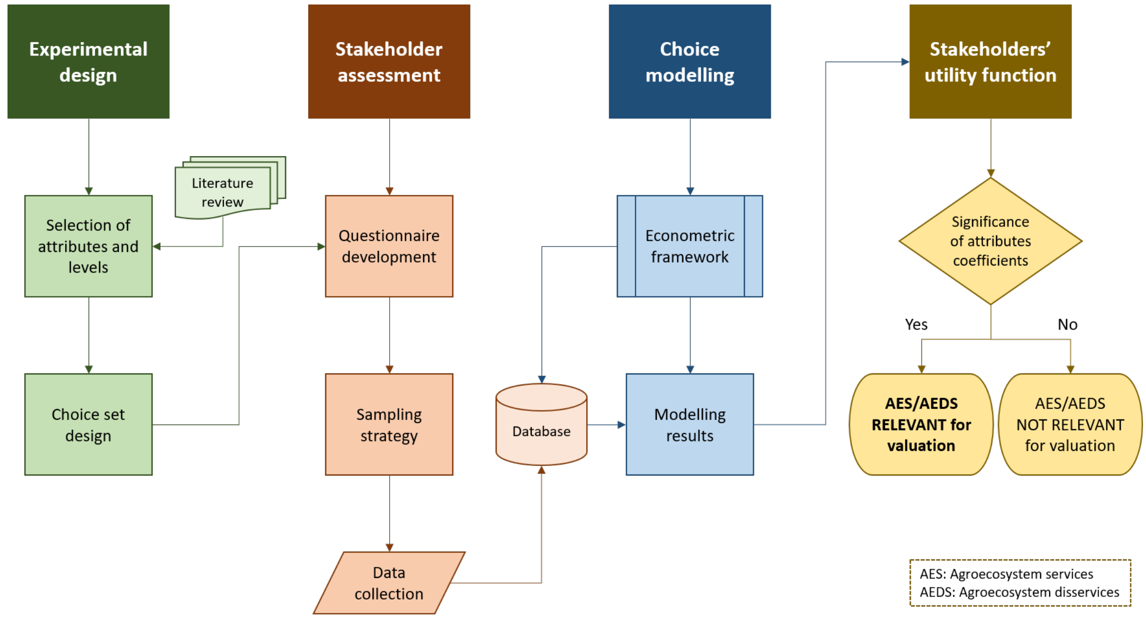


Figure 2.3. Process followed to implement the choice experiment method.

### *2.3.2.1. Experimental design*

The attributes included in the choice experiment design are associated with the relevant AES and AEDS identified in Section 2.2 (Table 2.3). The indicators for the attributes were selected following those proposed by Maes et al. (2016) and the van Oudenhoven et al. (2018) criteria. The selection of attribute levels was based on a review of available literature about the agroecosystems in the case study area (e.g. Alcon et al., 2017; Perni and Martínez-Paz, 2017; Albaladejo-García et al., 2020; Martin-Gorriz et al., 2020a), and focused on the specific indicators selected for the attributes. These attribute levels were chosen to cover the range of AES and AEDS of the three main agroecosystems included in the case study: rainfed, traditionally irrigated, and intensively irrigated. Hence, each level of every attribute represented a different agroecosystem in the case study. Levels were measured in physical or monetary units to improve the reliability of the experiment.

The attributes for the provisioning AES and AEDS were associated with the value of crop production (yield) and irrigation water use (water supply for irrigation). Economic yield was selected as an indicator to replace food supply, thereby homogenising different crop yields. The yield indicator levels were obtained from Alcon et al. (2017) and Lehtonen et al. (2020). Food provision can be easily evaluated as an AES since food can be exchanged in a market. However, the economic value of this AES goes beyond the market value of agricultural production because it comprises the contribution of agriculture to food security (option value).

Water management is a crucial issue within the case study area, not only for farmers and policy makers, but for the entire society due to competition for water among sectors (Perni and

Martínez-Paz, 2017; Zabala et al., 2019). The selected indicator was therefore the amount of water employed directly for irrigation, and the levels were obtained from Alcon et al. (2017). This means that when fresh water, which serves as a limiting resource in water-scarce regions, is directly employed in agriculture, it is not available for alternative uses in other ecosystems. Thus, the greater the use of irrigation water, the greater the AEDS provided by the agroecosystems.

The attributes related to the regulating AES and AEDS were carbon balance, temperature regulation, groundwater pollution, erosion, bird richness, and resilience. Carbon balance is defined as the net uptake of greenhouse gases by agroecosystems, and its indicator summarises the difference between carbon sequestration by, and greenhouse gas emissions from the agroecosystems. Therefore, carbon balance represents both the AES related to global climate regulation and the AEDS related to the emission of contaminants into the atmosphere. Hence, positive values of this indicator are associated with regulation of AES. The levels of this attribute were obtained from carbon balance data for the main crops grown in the case study area, following Martin-Gorriz et al. (2020a, 2020b).

Another of the AES, climate regulation, was considered because it is influenced by agricultural practices (Almagro et al., 2016). Irrigated agriculture can reduce the local temperature (Albaladejo-García et al., 2020), and is, therefore, expected to have a positive impact on human wellbeing in semiarid areas. Therefore, temperature regulation was included in the experimental design as an indicator for the local climate regulation AES. Attribute levels of local climate regulation were obtained from Albaladejo-García et al. (2020), who suggested that irrigated agroecosystems may reduce the land surface temperature by up to 2 °C compared to rainfed agroecosystems.

Groundwater pollution is a growing phenomenon in Mediterranean regions and is mainly caused by diffuse pollution from agriculture (Alcolea et al., 2019). It reveals how agroecosystems may negatively impact other ecosystems and represents the AEDS associated with water purification and waste treatment. The indicator used to measure groundwater pollution was the nitrate concentration in aquifers associated with each of the agroecosystems in the Region of Murcia (CHS, 2017).

Conventional agricultural practices, such as regular tillage and herbicide treatments, tend to erode soil (Montgomery, 2007). The negative contribution of agriculture to soil maintenance can, therefore, be seen as a disservice originating from agroecosystems, and annual erosion rates could be used as an indicator. However, given the great variety of erosion rates among agroecosystems, and even within each of the agroecosystems (García-Ruiz et al., 2013) included in the case study, this indicator was finally translated into a dummy variable which distinguished between high and low levels.

Contributions of the case study agroecosystems to biodiversity were measured as bird richness. Selection of this indicator was motivated by the fact that bird richness in the area had declined prior to this study as a consequence of agricultural activity (Palacín and Alonso, 2018; Martínez-López et al., 2019). In addition, bird richness is an easily understood indicator, as shown in several agroecosystem valuations (Rodríguez-Entrena et al., 2014; Varela et al., 2018). Biodiversity levels were defined as the share of the potential of bird richness that could be found in each of the agroecosystems, following the Perni and Martínez-Paz (2017) procedure. Bird species richness is known to be enhanced by crop diversity and heterogeneous landscapes (Stjernman et al., 2019), even in woody crops (Rime et al., 2020). Therefore, low intensity agroecosystems together with heterogeneous landscapes, such as the traditionally irrigated agroecosystem in the case study, were expected to provide greater bird species richness. Similarly, it was expected that the intensively irrigated agroecosystem, dominated by monoculture, would present a 60% bird richness level with respect to the potential, whereas for the rainfed agroecosystem, with low intensity agriculture and homogeneous landscapes, a value of 80% was expected. Although the contribution of agroecosystems to biodiversity is not always linear according to agricultural intensity (Beckmann et al., 2019), in this study it was assumed that the less intensive the agriculture, the greater the biodiversity the agroecosystem would hold, hence, the more resilient it would be (Augeraud-Véron et al., 2019). Consequently, rainfed and traditionally irrigated agroecosystems were assumed to be more resilient than the intensively irrigated ones. The final regulating attribute, resilience, was measured as the capacity of the agroecosystems to adapt to climate change. Given the difficulty of summarising this concept in just one measurable indicator (Cabell and Oelofse, 2012), it was included as a dummy variable based on whether the agroecosystem could adapt to climate change or not.

The cultural contribution of agroecosystems to human wellbeing was included in the experiment as four attributes. Agroecosystems contribute to the traditions and cultural identity of agricultural areas as well as providing landscapes for visual enjoyment and environments for leisure and recreational activities. The contribution to cognitive development and good living was included as the generation of local employment (Laterra et al., 2019). The capacity of each agroecosystem to generate employment was measured as the number of hours of labour needed to manage the agroecosystem, obtained from Alcon et al. (2017) and Lehtonen et al. (2020).

Table 2.3. Attributes and levels in the choice experiment.

| | Agroecosystem (AES/AEDS) | Attribute (CODE) | Definition (Indicator) | Units | Levels |
|---|---|---|---|---|---|
| Provisioning services | Food (AES) | Yield (FOOD) | Annual incomes received by farmers | €/ha/year | < 5,000*<br>5,000 - 15,000<br>> 15,000 |
| | Irrigation water (AEDS) | Water supply for irrigation (WATER) | Irrigation water supplied to crop system | $m^3$/ha/year | < 3,000*<br>3,000-5,000<br>> 5,000 |
| Regulating services | Emissions of contaminants to the atmosphere (AEDS)<br>Global climate regulation (AES) | Carbon balance (CARBON) | Net balance between $CO_{2eq}$ sequestration and emission | tonnes $CO_{2eq}$/ha/year | < 15*<br>15-30<br>> 30 |
| | Local climate regulation (AES) | Temperature regulation (TEMPE) | Temperature changes on the land surface | ºC | 0*<br>-1 ºC<br>-2 ºC |
| | Water purification and waste treatment (AEDS) | Groundwater pollution (POLL) | Nitrate concentration in aquifers | mg $NO_3^-$/L | < 50*<br>50-200<br>> 200 |
| | Soil maintenance (AES/AEDS) | Erosion (EROS) | Loss of soil due to wind or precipitation | - | High*<br>Low |
| | Biodiversity (AES/AEDS) | Bird species richness (BIOD) | Bird species richness with respect to potential | % | 100 %<br>80 %*<br>60 % |
| | Resilience (AES) | Resilience (RESL) | Agroecosystem's climate change adaptation | - | High*<br>Low |
| Cultural services | Culture, art and design (AES) | Cultural heritage (CHERIT) | Presence of cultural elements linked to agriculture | - | No*<br>Yes |
| | Aesthetic values (AES) | Landscape (LAND) | Scenic landscape beauty | - | Rainfed agroecosystem*<br>Traditional irrigated agroecosystem<br>Highly-intensive irrigated agroecosystem |
| | Opportunities for recreation and tourism (AES) | Recreation and tourism (RECRE) | Chance of enjoying activities in agroecosystems | - | No*<br>Yes |
| | Cognitive development and good living (AES) | Employment generation (EMPGE) | Labour related to agroecosystems management | hours/ha/year | < 100*<br>100-500<br>> 500 |

*Attribute levels which comprise the SQ alternative (rainfed agroecosystem)

In the experimental design, the attribute levels were combined by applying an S-efficiency design, using the Ngene 1.0.2 software package (Rose et al., 2010). The S-efficiency design was chosen for this study to minimise the sample size requirements (Rose and Bliemer, 2013) because of the small target population: the agroecosystem stakeholders in the Region of Murcia. The final design comprised 18 choice sets grouped into 3 blocks, which were randomly assigned to the stakeholders. Hence, each stakeholder was presented with six choice sets consisting of three alternatives each, which represented the different agroecosystems included in the case study: one alternative was the rainfed agroecosystem, used as the SQ alternative, and the other alternatives represented the irrigated agroecosystems. An example of a choice set is provided in Appendix 2.A (Figure 2.A.1).

#### *2.3.2.2. Sampling and data collection: Stakeholder assessment*

The AES and AEDS significant for agroecosystem valuation were assessed through face-to-face interviews with agroecosystem stakeholders of the Region of Murcia (the target population). Agroecosystem stakeholders included farmers, agricultural technicians, irrigation community managers, agricultural R&D managers in private companies, members of scientific bodies such as universities and research institutes, public administrators, and local communities involved in agriculture (Alcon et al., 2014). Therefore, the stakeholders comprised any group or individual affecting or affected by the AES and AEDS (Hein et al., 2006). In accordance with this definition, relevant institutions and individuals involved in agricultural decision-making were identified and asked to participate. Thus, an initial selection of 10 stakeholders was identified and contacted for interviewing. Once the interviews began, a snowball sampling method was followed to select other relevant stakeholders (Biernacki and Waldorf, 1981; Reed et al., 2009). In total, 44 agroecosystem stakeholders were successfully interviewed and classified into four key groups:

- Users (11): This group included farmers and technicians who worked directly in the agroecosystems.
- Researchers (10): This group comprised agronomic engineers, scientists, and economists who conducted research in the case study agroecosystems.
- Public managers (13): This group included managers from regional and national organisations responsible for water use and agricultural land management.
- Civil society (10): This group comprised NGOs, labour unions, political parties, and other associations.

The stakeholder interviews were conducted face-to-face, between July and September 2018, based on a two-part questionnaire. The first part concerned stakeholder perceptions and attitudes about the AES and AEDS provided by agroecosystems in the Region of Murcia, and the second part comprised the choice experiment. The stakeholders were asked to choose the agricultural system they would like to implement in the Region of Murcia.

Despite the extensive use of choice experiments in environmental economics, some limitations continue to arise from the employment of this method, mainly related to its hypothetical nature (Alemu and Olsen, 2018). The issue is whether the respondents' hypothetical choices would correspond to their actual behaviour if they faced similar choice situations in real life (Carlsson, 2010). Furthermore, the application of such a stated preference method, including environmental goods and services that are complex and unfamiliar to respondents, has been criticised on the basis that respondents cannot give accurate responses as their preferences are not fully discovered (Braga and Starmer, 2005). To mitigate these possible limitations, certain factors were taken into consideration in relation to sampling and data collection. The group of agroecosystem stakeholders interviewed comprised experts in their respective fields of agricultural work; therefore, it helped to ensure they were familiar with the choice situations they faced. In addition, two ex-ante strategies were applied to mitigate hypothetical bias (Loomis, 2014). First, prior to participating in the choice experiment, the respondents were thoroughly informed about the AES and AEDS used, the attributes and levels included, and the purpose of the study, using cheap talk script (Champ et al., 2009). Second, they were advised that the survey results would be employed to inform agricultural policy makers and, therefore, they would have an influence on future agricultural policies in the Region of Murcia.

### *2.3.2.3. Econometric framework*

According to the random utility theory (McFadden, 1974), the utility $U_{ij}$ for an individual *i* provided by an agroecosystem alternative *j* can be decomposed into a deterministic ($V_{ij}$) and a stochastic part ($\varepsilon_{ij}$), considered additively:

$$U_{ij} = V_{ij} + \varepsilon_{ij} = \sum_{k=1}^{K} \beta_k X_{ikj} + \varepsilon_{ij} \qquad k = 1, \ldots, K \tag{2.1}$$

where $V_{ij}$ represents the observed elements of the utility determined by the *k* attribute levels ($X_{ikj}$), and $\varepsilon_{ij}$ is a random error with an independent and identically distributed extreme-value distribution (Train, 2009). Assuming a linear relationship among the attribute levels, $\beta_k$ is the individual marginal utility obtained from each *k* attribute, reflecting how the utility level changes if the provision of AES and AEDS increases.

The agroecosystem alternatives chosen by the respondents allow us to explore the probability of choosing an alternative *j* and to estimate the marginal utilities, $\beta_{ik}$, which maximise it. The conditional logit (CL) model (Train, 2009) is widely used to estimate the probabilities of such choices. Nevertheless, the CL model implies some restrictive assumptions (no random taste variation, restrictive substitution patterns, and no correlation of unobserved factors), the most relevant being the independence of irrelevant alternatives (IIA), which assumes that the probability of choosing an alternative is not influenced by the existence of any other alternatives. The IIA principle can be contrasted by the Hausman test (Hausman and McFadden, 1984). If the null hypothesis of the Hausman test is not rejected, the CL model is a suitable to estimate the stakeholders' utility function. However, if it is rejected, another specification model, such as the mixed logit model, should be employed instead.

A linear specification was employed to estimate the utility function. All attributes were assumed to be continuous variables, except *CHERIT*, *RECRE*, *LAND*, and RESL, which were assumed to be categorical. Preference heterogeneity was examined using the interactions between some attributes and the stakeholders' type. A positive sign for coefficients related to AES and a negative sign for those referring to AEDS were expected, but it was rather difficult to hypothesise which AES and AEDS would have a significant role in the explanation of the stakeholders' choices.

The results from the CL model were also used to calculate the relative importance or weight of each attribute. Adapting the Danner et al. (2017) procedure for continuous attributes, the relative importance of attribute $k$ ($RI_k$) was calculated as follows:

$$RI_k = \frac{|\beta_k \bar{X}_k|}{\sum_{k=1}^{K} |\beta_k \bar{X}_k|} \tag{2.2}$$

where $\bar{X}_k$ represents the average of the attribute $k$, and $RI_k$ represents, therefore, the relative contribution of the attribute $k$ to the total utility, evaluated for the average value of each attribute.

## 2.4. Results

The stakeholders' preferences were analysed using two CL models (Table 2.4). The CL model specification was appropriate since the Hausman test results (HT) validated the existence of IIA (HT = 10.03; $\chi^2_{0.05;11}$ = 19.675). Both Model 1 and Model 2 were based on the main-effects CL model, and Model 2 included stakeholder-group interactions. Significant differences between Model 1 and Model 2 were found with the log-likelihood ratio (LR) test (LR test = 31.234; $\chi^2_{0.05;3}$ = 7.815). Moreover, the accuracy of the choice models, which refers to the ability of both models to explain stakeholders' preferences in a precise manner, was evaluated through the pseudo-$R^2$ and the percent correctly predicted (PCC). Again, Model 2 performed better than Model 1 in terms of the pseudo-$R^2$, PCC, AIC, and BIC criteria; thus, it was used as the basis for further discussion of the results.

The Model 2 results showed a significant negative coefficient for the rainfed agroecosystem (SQ) alternative ($p < 0.01$), reflecting the disutility provided by this agroecosystem in terms of the AES and AEDS provided. The interaction terms between the SQ alternative and the stakeholder groups were significant, showing the heterogeneity of preferences among the stakeholder groups regarding the rainfed agroecosystem: the farmers perceived the highest disutility from the rainfed agroecosystem, followed by the researchers, public managers, and civil society.

Table 2.4. Stakeholders' utility function. Estimated CL models.

| | Model 1 | | | Model 2 | | |
|---|---|---|---|---|---|---|
| | CL model | | | CL model and stakeholder heterogeneity | | |
| | Coef. | | Std. Err. | Coef. | | Std. Err. |
| SQ | -0.874 | | 0.812 | -2.889 | *** | 0.987 |
| FOOD | $3.32 \cdot 10^{-5}$ | ** | $1.58 \cdot 10^{-5}$ | $3.21 \cdot 10^{-5}$ | ** | $1.61 \cdot 10^{-5}$ |
| WATER | $-2.23 \cdot 10^{-4}$ | * | $1.28 \cdot 10^{-5}$ | $-2.33 \cdot 10^{-4}$ | * | $1.32 \cdot 10^{-4}$ |
| CARBON | 0.017 | | 0.011 | 0.016 | | 0.011 |
| TEMPE | 0.406 | *** | 0.150 | 0.423 | *** | 0.155 |
| POLL | -0.009 | *** | 0.001 | -0.009 | *** | 0.002 |
| EROS | -0.250 | | 0.260 | -0.182 | | 0.267 |
| BIOD | 0.018 | ** | 0.009 | 0.018 | ** | 0.009 |
| RESIL | 0.120 | | 0.285 | 0.04 | | 0.290 |
| CHERIT | -0.083 | | 0.254 | -0.101 | | 0.263 |
| LAND | 0.349 | | 0.257 | 0.349 | | 0.264 |
| RECRE | 0.835 | *** | 0.254 | 0.763 | *** | 0.260 |
| EMPGE | $2.46 \cdot 10^{-5}$ | | 0.001 | $-1.20 \cdot 10^{-4}$ | | 0.001 |
| RESEARCHER*SQ | | | | 1.721 | *** | 0.617 |
| PUBLIC MANAGER*SQ | | | | 2.161 | *** | 0.577 |
| SOCIETY*SQ | | | | 2.696 | *** | 0.597 |
| Number of observations | 792 | | | 792 | | |
| Log likelihood | -243.456 | | | -227.839 | | |
| Pseudo $R^2$ | 0.161 | | | 0.214 | | |
| PCC (%) | 70.960 | | | 74.490 | | |
| AIC | 512.911 | | | 487.677 | | |
| BIC | 573.681 | | | 562.470 | | |

Analysis of the significance of the Model 2 coefficients to determine the AES and AEDS, which really explain the stakeholders' utility function, showed that the valuation of 6 out of 12 AES and AEDS was relevant. Of the provisioning services, the yield from agricultural activities (FOOD) ($p < 0.05$) and water supply for irrigation (WATER) ($p < 0.1$) explained the stakeholders' choices in terms of AES and AEDS, respectively. The positive sign of the FOOD coefficient indicated that higher farm yield levels were preferred by the stakeholders. The negative sign of the WATER coefficient confirmed the disutility of this attribute, indicating its relevance among the AEDS.

Regarding the regulation of AES and AEDS, significant coefficients were found for the attributes: temperature regulation (TEMP) and groundwater pollution (POLL) ($p < 0.01$) as well as for the agroecosystem contribution to biodiversity maintenance (BIOD) ($p < 0.05$). Therefore, local climate regulation is seen as an AES that can mitigate high temperatures in semiarid areas. The negative sign of the POLL coefficient reflected the stakeholders' concerns about the environmental impact of agriculture in terms of pollution externalities, confirming the consideration of POLL as one of the AEDS. Bird richness is assumed to be a good indicator of biodiversity and should therefore be relevant among the AES provided by agroecosystems. The positive BIOD coefficient reflected the utility perceived by stakeholders for the enhancement of biodiversity within agricultural landscapes. The non-significant coefficients obtained for erosion (EROS) and carbon balance (CARBON) indicated their irrelevance to the stakeholders.

Despite the fact that in the literature there are many references to the impacts of cultural AES on human wellbeing, in agroecosystems, only recreation and leisure (RECRE) showed a significant effect ($p < 0.05$) on the stakeholders' utility function. Although the agricultural contribution to direct employment was expected to be significant due to the socioeconomic idiosyncrasy of the case study agroecosystems, the EMPGE attribute coefficient was not significant.

Therefore, according to the stakeholder preferences, two provisioning, three regulating, and one cultural AES and AEDS were identified as worthy of valuation, due to their notable impact on human wellbeing. Moreover, the coefficient sign reflected the positive or negative contribution to social welfare and verified our previous consideration of the attributes as either AES or AEDS. The coefficient signs for WATER and POLL were negative, corroborating their definition as AEDS, thus reflecting the disutility associated with higher attribute levels. The remaining significant AES (FOOD, TEMPE, RECRE, and BIOD) had positive coefficient signs, showing that they were considered as AES by the stakeholders.

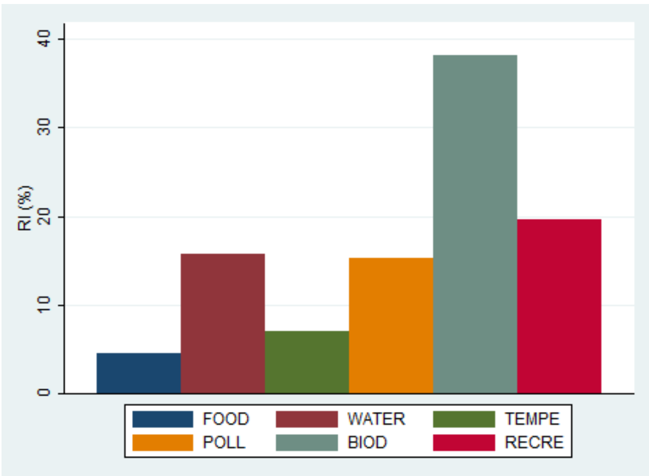


Figure 2.4. Relative importance (RI) of the AES/AEDS for valuation.

The results also enabled calculation of the relative importance of each of the significant AES and AEDS in the agroecosystem valuation (Figure 2.4). The stakeholders' choices revealed biodiversity (38%) as the most important of the AES to be valued, followed by recreation (20%), temperature regulation (7%), and food provision (5%). Among the AEDS, water supply for irrigation and groundwater pollution were considered of equal weight (at 15% each).

## 2.5. Discussion

The analysis of stakeholders' preferences for AES and AEDS has been used to validate a comprehensive approach for the valuation of the AES and AEDS provided by semiarid western Mediterranean agroecosystems. This approach is based on the framework for anthropised ecosystems developed by Barot et al. (2017), and it adapted the main accepted ecosystem services paradigms: MEA (2005), TEEB (2010), and CICES (Haines-Young and Potschin, 2018) to the particular case of agroecosystems. The stakeholder assessment enabled us to determine which AES and AEDS should be relevant for an agroecosystem valuation. Using choice experiments in the context of stakeholder assessment and AES and AEDS valuation, we considered the perceived trade-offs between AES and AEDS in an integrated way. The approach included at least one of the AES or AEDS from every category (provisioning, regulating, and cultural), in line with the multifunctional character of agricultural activity (Huang et al., 2015).

The approach presented here integrated AES and AEDS into a common valuation framework. The results from the stakeholder assessment revealed that the valuation of agroecosystems needs to deal with both positive and negative outcomes. Hence, negative contributions to human wellbeing should be included when the aim is to value the overall impact of agriculture on wellbeing. Indeed, Figure 2.4 shows that the relative importance that stakeholders attached to AEDS is 30% of the total. These results reinforce the claims of Schaubroeck (2017) and Blanco et al. (2019), who suggested an equal consideration of services and disservices, not only in economic valuation but also in research and policy agendas. Ignoring the values of AEDS when assessing agroecosystem contributions to human wellbeing could lead to overestimation of the benefits provided and thus to incorrect policy decisions, due to the undervaluation of costs. The holistic valuation of AES and AEDS could enable more efficient allocation of economic resources because it could be more cost-effective to mitigate disservices than to increase services (Shackleton et al., 2016). Consequently, these results could be used as a guide to improve our knowledge about the relative values that societies place on AES and AEDS.

The proposed approach also endorses the integration of provisioning and non-provisioning services, which traditionally have been considered separately in economic valuation (distinguished as marketed and non-marketed services). The economic valuation of agroecosystems could be developed either according to their capacity to provide services and disservices (supply-side valuation), or considering the social demand for the services and disservices provided by the agroecosystems (demand-side valuation). Supply-side valuation involves cost-based and production-based methods, which usually integrate all the types of services and disservices provided (Martín-López et al., 2014), whereas demand-side valuation involves preference-based approaches and, consequently, focuses on non-marketed services and disservices (Niedermayr et al., 2018). Thus, our results revealed that, although non-provisioning services were dominant (Figure 2.4), agroecosystem valuation needs to consider both services and disservices, consistent with the ongoing discussion in the literature (Bernués et al., 2019).

Considering all the AES and AEDS, it is not surprising that both provisioning services and disservices have been shown to be valuation relevant. Agroecosystems are ecosystems created by humans to provide food, therefore, the significance of provisioning services must be valued. However, in the present study, the relative importance of provisioning services in relation to the overall AES and AEDS to be valued was not as high as expected. Provisioning services represented approximately 20% of the total importance (Figure 2.4), in line with the findings of Bernués et al. (2019) for the Mediterranean region. This evidence indicates that the value of an agroecosystem goes beyond the direct use that the socioeconomic system obtains from it.

Regulating AES and AEDS are essential for agroecosystem assessment due to the relevance of their contribution to wellbeing, as they generate environmental benefits and costs, respectively. The findings of this study indicated that the relevance of both regulating AES and AEDS was broadly recognised by the stakeholders who stated that the indirect use value of regulating AES and AEDS is key to human wellbeing and valuated their relative importance in the case study agroecosystems at approximately 60% (Figure 2.4). These results are consistent with Bernués et al. (2019), who found that the indirect use value in a Mediterranean agroecosystem contributed 53.2% of the total value estimated. Of the regulating AES, climate regulation and maintenance of lifecycles and genetic diversity were relevant to the valuation. Temperature regulation, an indicator of local climate regulation, was rated as important by stakeholders in our study as it has a great influence on wellbeing; however, no significant effect was found for the indicator of global climate regulation. These findings may be related to the warm weather in the study area which is more easily perceived by stakeholders than climate change effects. This explanation is supported by Olander et al. (2017), who stated that people tend to value benefits that provide more direct and closer effects. Biodiversity was by far the most important of the regulating AES valued (Figure 2.4) in our study. The value of biodiversity comes from a great variety of sources (Paul et al., 2020), beyond its indirect use or option values. In addition, benefits can be obtained from biodiversity due to the positive effects it may have on human health (Sandifer et al., 2015), mainly through the emotional and psychological aspects of human wellbeing (Fuller et al., 2007; Dallimer et al., 2012). Regarding soil erosion, even though this regulating factor is important in certain agroecosystems, such as rainfed Mediterranean (Almagro et al., 2016) or diversified (Alcon et al., 2020) agroecosystems, it was not considered relevant by the stakeholders who participated in the present study. Many of the consulted stakeholders, particularly in the Users and Civil society groups, did not consider soil erosion a major concern in the case study agroecosystems. This finding is supported by Cerdá et al. (2018), who determined that a number of citrus farmers did not consider soil erosion a problem in southeast Spain. However, the perceived lack of importance of soil erosion may be related to a lack of environmental education and awareness regarding soil erosion and conservation issues (Oñate and Peco, 2005; Sastre et al., 2017). A similar statement could be made about resilience. Despite its noticeable importance in guaranteeing agricultural sustainability under natural hazards and climate change (Peterson et al., 2018), stakeholder awareness of the negative impacts of climate change appears to be lacking (Esteve et al., 2018).

The cultural services provided by agroecosystems have social benefits generally associated with the use and enjoyment of these environments. Our results showed that, among the cultural AES valuated, leisure and recreation were perceived to have significant influences on wellbeing greater than those of the landscape, cultural heritage, and cognitive development. In fact, leisure and recreation is considered the broadest service and can partially encompass other cultural AES (García-Llorente et al., 2012). In this case study, to a certain extent, attributes such as the landscape and cultural heritage linked to the agroecosystems and their relative importance approached 20%, contrasting with Bernués et al. (2019) who found that cultural services represented 8% of the overall demand for AES. However, Martínez-Paz et al. (2019) determined that cultural services accounted for 42% of the relative importance of the AES provided by the Huerta of the Region of Murcia, a specific agroecosystem located within the traditionally irrigated agroecosystem in the case study area. These differences in relative importance of cultural services again show the importance of contextual background for understanding the results obtained from the valuation of AES and AEDS.

Water management is crucial for semiarid Mediterranean farming. The Mediterranean area in general, and the case study area in particular, are characterised by a semiarid climate, which makes dealing with water scarcity one of the main challenges in these agroecosystems. This fact was reflected in the stakeholder utility function as the only two significant AEDS in the explanation of the stakeholder choices were related to water management. Supplying water for irrigation is perceived to cause a reduction in available water resources, which are indeed limited. This could imply, in turn, an opportunity cost, because alternative uses of water show higher water-productivity values. The rivalry associated with competing uses of water resources and the social dilemma of supplying reclaimed water to competing ecosystems, is highlighted by Zabala et al. (2019) in their Region of Murcia case study. Moreover, these concerns refer not only to water employed as an input to agroecosystems, but also to water flows supplied by agroecosystems. Either frequent or excessive nitrogen fertilisation could have negative consequences for the agroecosystem and surrounding ecosystems. Aquifers are particularly affected by agroecosystem nitrate leaching and runoff. Therefore, the recognition by stakeholders that agroecosystems contribute negatively to water purification and waste treatment evidences the negative externality supplied by agricultural activity, which would be mitigated only if wastewater coming from agricultural systems could be properly treated (Sepehri and Sarrafzadeh, 2018; Sepehri et al., 2020). This implies not only evident environmental costs, but also economic and social costs, especially when nitrate runoff reaches high-value ecosystems, as occurs in the case study area where the Mar Menor coastal lagoon is impacted (Velasco et al., 2018). Hence, the joint consideration of AES and AEDS for water management seems to be a key element in semiarid Mediterranean agroecosystem valuation.

The implications of the findings of this case study could be applied to improve agricultural policy design. Policy makers need to boost the provision of the most relevant AES, while mitigating AEDS, in accordance with case-specific agroecosystems and their surrounding areas. Increasing human wellbeing in semiarid western Mediterranean agroecosystems implies the enhancement of food provision, local climate regulation, biodiversity, and recreational activities within the agroecosystems. In addition, this should be supported with measures or strategies focused on reducing the water supply for irrigation, such as regulated deficit irrigation, and mitigation of diffuse pollution from agricultural systems (Alcon et al., 2021). We note that the findings of the present study could not be used as a tool to publicly support the transition from other land uses to agroecosystems (e.g. to support forest conversion) because land use changes, which may imply the transition from or to other ecosystem types, were not considered in our agroecosystem assessment approach. The integration of land use changes and ecosystem services and disservices, where agroecosystems might play a key role, should be considered in future research.

## 2.6. Conclusions

Determination of the most relevant AES and AEDS for valuation was the main motivation of this study. To accomplish this, it was necessary to adapt the existing ecosystem services paradigms to the particular case of the agroecosystem. Therefore, a comprehensive approach for AES and AEDS was proposed and validated by stakeholder assessment. Determining the stakeholder preferences enabled us to establish the AES and AEDS that semiarid western

Mediterranean agroecosystem valuation should include: food provision and fresh water (as provisioning services), local climate regulation and wastewater treatment (as regulating services), the contribution to recreation and tourism (as cultural services), and biodiversity.

Regarding management implications, the results indicated that an increase in human wellbeing comes from the following: promotion of agricultural and natural resources, policies that maximise the agricultural contribution to food provision, reduction of the water supply for irrigation, lowering of the local temperature, minimising groundwater pollution, creation of an environment that supports recreational and leisure activities, and encouraging biodiversity conservation. Therefore, this comprehensive approach serves to raise awareness of the need to consider AES and AEDS holistically in agri-environmental policy design. This approach will be a key tool for forthcoming agroecosystem economic valuations, which will translate the social demand for AES and AEDS into monetary terms, and will ensure efficiency in the design of socially acceptable agri-environmental schemes.

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# Appendix 2.A. Figures

| Agroecosystem 1.1. | Rainfed (SQ) | Irrigated 1 | Irrigated 2 |
|---|---|---|---|
| Yield (€/ha/year) | < 5.000 | 5.000 – 15.000 | < 5.000 |
| Water supply for irrigation ($m^3$/ha/year) | < 3.000 | 3.000 – 5.000 | > 5.000 |
| Carbon balance (tonnes $CO_{2eq}$/ha/year) | < 15 | 15 - 30 | < 15 |
| Temperature regulation (ºC) | | | 1 ºC |
| Groundwater pollution (mg $NO_3^-$/L) | < 50 | 50 - 200 | > 200 |
| Erosion | High | High | High |
| Bird species richness (%) | 80 % | 100 % | 80 % |
| Resilience | High | Low | High |
| Cultural heritage | No | Yes | Yes |
| Landscape | Rainfed | Highly-intensive | Traditional |
| Recreation and leisure | No | Yes | Yes |
| Employment generation (hours/ha/year) | < 100 | < 100 | 100 - 500 |
| **Choose an alternative** | | | |

Figure 2.A.1. Example of a choice set

# Chapter 3. Integrated valuation of semiarid Mediterranean agroecosystem services and disservices

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## Highlights

- A comprehensive valuation of agroecosystem services and disservices is provided.
- Non-linear preferences for provisioning and regulating services are identified.
- Omitting disservices leads to overestimation of the agroecosystem value.
- Agricultural policy should deal with the mitigation of agroecosystem disservices.

## Abstract

Agroecosystems are anthropised ecosystems where human activities, mainly agricultural practices, affect the innate functioning, leading to the provision of agroecosystem services (AES) and disservices (AEDS). This study presents a novel and integrated economic valuation of the AES and AEDS provided in a water-scarce Mediterranean area (south-eastern Spain), using a discrete choice experiment. The results reveal the social demand for AES and the disutility of AEDS, as well as the non-linearity in marginal utility for some of these AES and AEDS. Food provision, temperature regulation, leisure and recreation and biodiversity are socially perceived as AES. The water supply for irrigation switches between AES and AEDS depending on its provision level, while groundwater pollution is conceived as one of the AEDS. The integrated non-market value of AES and AEDS reaches 794 €/ha/year for the entire agroecosystem. This work provides guidelines for policy makers in the design of socially supported agricultural policies.

Keywords: Discrete choice experiments; Non-Market valuation; Non-Linear preferences; Trade-offs; Wellbeing.

## 3.1. Introduction

Today, it is well-known that agriculture produces more than just food and fibre. The contribution of agriculture to society also encompasses the provision of non-commodity goods and services such as soil erosion control, climate regulation and biodiversity maintenance. Agroecosystem services (AES), defined as the contribution of an agroecosystem to human wellbeing (TEEB, 2010), represent an appropriate paradigm to embrace all these agricultural outputs. The ecosystem service framework, first developed by the Millennium Ecosystem Assessment (MEA, 2005) and extended by TEEB (2010) and Haines-Young and Potschin (2018), constitutes an increasingly-used tool for assessment of the agricultural impacts on human wellbeing. This approach, which considers the multifunctionality of agriculture (Huang et al., 2015), covers the multiple agricultural outputs: both private and public goods and services (Fisher et al., 2009; Cooper et al., 2009). Essentially, the ecosystem service framework allows the assessment of the agroecosystems benefits for human wellbeing.

However, despite its apparent simplicity, this framework becomes complex when applied to agroecosystems, due to, among other factors, the presence of agroecosystem disservices (AEDS) and their trade-offs with AES. The concept of AEDS, defined as “the ecosystem generated functions, processes and attributes that result in perceived or actual negative impacts on human wellbeing” (Shackleton et al., 2016), reveals that some agricultural contributions to human wellbeing could be non-positive. In fact, agriculture is responsible for more than 70% of the annual consumption of water resources worldwide (WWAP, 2016), occupying nearly 50% of the Earth’s land surface and emitting around 25% of the global anthropogenic greenhouse gases output (IPCC, 2019). Overlooking the existence of AEDS could have problematic consequences for research and policy orientations (Shackleton et al., 2016). Negative effects of agroecosystems on other ecosystems do exist (Power, 2010); thus, neglecting them implies not recognising a part of the overall contribution of agriculture to human wellbeing. Therefore, agricultural policy measures could be more cost-effective and efficient if they were to mitigate AEDS instead of enhancing AES (Shackleton et al., 2016).

Economic valuation of AES and AEDS allows recognition of their relative importance and, above all, the orientation of policy decisions when considering the overall contribution of agroecosystems to human wellbeing (de Groot et al., 2012). The implementation of sustainable agricultural practices implies the joint consideration of both the supply and demand of AES and AEDS. While the supply of AES and AEDS considers farmers’ practices, their demand should be analysed regarding social preferences. The valuation of some AES, such as food provision, is straightforward due to the existence of markets. However, most AES and AEDS are non-marketed and require alternative methods to estimate their values. Stated preference methods, such as contingent valuation and discrete choice experiments (DCEs), are the methods used most for this purpose, since they allow estimation of the social demand for their provision, and thus the willingness to pay for, and the value of, AES and AEDS variations (TEEB, 2018). These methods can also be combined with multi-criteria analysis to assess the provision of information to individuals and its impact (Martin-Ortega and Berbel, 2010).

In such a context, the need for an integrated framework for the valuation of AES and AEDS should be addressed from the demand side. Specifically, the present work aims to value economically the integrated provision of AES and AEDS in a water-scarce Mediterranean agroecosystem (south-eastern Spain), using a DCE. Our results are expected to have policy relevance as they provide a direct estimation of social preferences regarding the contribution of agriculture to human wellbeing. They will also guide the design of socially accepted agri-environment policies, bearing in mind the next reform of the Common Agricultural Policy of the European Union (Pe'er et al., 2019).

Previous studies have analysed the interrelationships between AES and AEDS by using different approaches. Zhang et al. (2007) were the first authors who recognised the presence of AEDS in their assessment scenario. However, their work focused more broadly on the AEDS to agriculture; that is, the AEDS provided by other ecosystems to agriculture. Power (2010) assessed AEDS both to and from agriculture. He highlighted nutrient loss, pollution and emission of greenhouse gases as the main AEDS. Ango et al. (2014) analysed farmers' management of trees in agricultural landscapes in Ethiopia, considering their provision of AES and AEDS, while Ma et al. (2015) assessed the AES and AEDS provided by a high-production agroecosystem in China, using emergy valuation. Nevertheless, it is difficult to find in the literature studies which quantify economically AES and AEDS in an integrated way (Chang et al., 2011; Hardaker et al., 2020; Sandhu et al., 2020). Therefore, to the best of our knowledge, this is the first work which attempts to value AEDS, and it provides an integrated non-market valuation of all agricultural contributions to human wellbeing by using stated preference methods.

The paper is structured as follows. In the following section the adequacy of the integrated valuation of both AES and AEDS is discussed in more depth. Section 3.3 explains the methodology employed, with special attention to the selection of AES and AEDS for the assessment, while Section 3.4 presents the main results. The discussion of the results, as well as their theoretical implications and policy applications, is provided in Section 3.5. Finally, in Section 3.6 the conclusions are stated.

## 3.2. The need for a framework for the integrated valuation of AES and AEDS

Since the publication of the MEA (2005), there has been a growing body of work aimed at measuring the benefits provided by agroecosystems (e.g. Granado-Díaz et al., 2019; Tienhaara et al., 2020). However, little is known about the negative effects, or social cost, of agroecosystems. AEDS is not a straightforward concept. Wellbeing reductions could be caused by both reducing AES and providing AEDS (Vaz et al., 2017). Due to the synergies among socio-ecological processes, AES and AEDS could vary simultaneously, providing trade-offs among them (Blanco et al., 2019). Anthropised ecosystems, such as agroecosystems, have to deal also with the effects of human management, which could both mitigate and promote the provision of AEDS (Barot et al., 2017). The same agroecosystem functions and processes could be perceived as AES or AEDS depending on people's behaviour, preferences and the socioeconomic context (Shackleton et al., 2016; Vaz et al., 2017).

Integrated assessment of AES and AEDS is needed for at least three reasons. Firstly, because considering only AES implies taking into account just part of the overall contribution of the agroecosystem to wellbeing (Schaubroeck, 2017). Secondly, the global assessment of AES and AEDS allows integration of the trade-offs between them and considers the net impact of agroecosystems on human wellbeing (Barot et al., 2017; Blanco et al., 2019). Finally, it helps to achieve better design of policies intended to produce sustainable and resilient agroecosystems (Sandhu et al., 2019). This becomes more significant when economic valuation is included in the assessment framework. If AEDS are ignored in policy design, this could lead to overestimation of the benefits provided by agroecosystems - which will be translated into suboptimal solutions - and could lead policy makers to make wrong decisions since they have not taken into account the costs they imply.

Recent literature makes the case for the introduction of AEDS into the assessment framework (e.g. Shackleton et al., 2016; Barot et al., 2017; Blanco et al., 2019; Sandhu et al., 2019). Specifically, Blanco et al. (2019) proposed that, to strengthen the inclusion of AEDS in research and policy analysis, it is necessary to: develop an AEDS classification which unifies further research; assess AES and AEDS in both biophysical and socio-economic terms, integrating them into a common framework; broaden the analysis to trade-offs among AEDS and between AES and AEDS; evaluate spatial and temporal variation in AEDS demand and supply; and integrate the assessment of AEDS co-production in research and policy agenda.

Although some steps have been taken to establish ecosystem disservices concepts and classification, there is still no widely accepted typology of disservices, which is especially notable if the focus is only on AEDS. Escobedo et al. (2011) split up disservices into three categories - financial costs, social nuisances and environmental pollution - to account for the effectiveness of urban forests in pollution mitigation. Von Döhren and Haase (2015) provided a literature review on ecosystem disservices research and clustered the negative effects of ecosystem functioning according to thematic fields: ecological, economic, health, psychological and general impacts on human wellbeing. This typology was also applied by Campagne et al. (2018) to assess the capacity of a French natural park to provide ecosystem services and disservices. On the other hand, Shackleton et al. (2016) defined six types of disservices according to the biotic or abiotic origin and the expected impact on the different aspects of human wellbeing: bio-economic, bio-health, bio-cultural, abiotic-economic, abiotic-health and abiotic-cultural. Following the approach of Shackleton et al. (2016), Vaz et al. (2017) established that disservices could be alternatively classified among health, material, security and safety, cultural and aesthetic and leisure and recreation typologies.

All these different proposals reveal that, despite the huge effort in defining a widely applicable classification of ecosystem disservices, greater consensus is needed to clarify and reach a widely accepted agreement on this matter. Moreover, controversy arises when the purpose is to integrate AES and AEDS in the same framework of assessment. The main accepted paradigms of ecosystem services, such as MEA (2005), TEEB (2010) and CICES (Haines-Young and Potschin, 2018), agree on the classification of services in provisioning, regulating and cultural categories for valuation purposes. However, it is a challenge to integrate AES in this categorisation, to link them with the established and proposed typologies of AEDS. This challenge might be overcome if similar classifications were applied to both AES and AEDS. Barot et al. (2017), who proposed a general framework to assess anthropised ecosystems, as well as Hardaker et al. (2020) and Sandhu et al. (2020), who provided integrated economic valuations of AES and AEDS, are a few examples of authors who claim to have applied the same existing categories for services to disservices.

The integrated assessment of AES and AEDS in both biophysical and socio-economic terms is gaining momentum in the literature (Blanco et al., 2019). Most of this research deals with some specific AEDS and how their flows impact on agroecosystems and socio-ecological systems. This is the case, for instance, of the work of Rasmussen et al. (2017), which assessed the switch between AES and AEDS that happens when wild animals and plants (biodiversity) are present excessively in agroecosystems, and of that of Pejchar et al. (2018), which showed the net effects of birds in agroecosystems. Other studies, however, have focused on empirical applications, from the analysis of trade-offs between AES and AEDS (Finney et al., 2017; Nguyen et al., 2018) to their integrated evaluation (Ma et al., 2015; Schäckermann et al., 2015; Shah et al., 2019; Blanco et al., 2020). In particular, Finney et al. (2017) showed how a mixture of cover crops provides AES and AEDS and how trade-offs among them arise, whilst

Nguyen et al. (2018) optimised the provision of AES and AEDS -particularly primary production, soil organic carbon, water use, nitrogen leaching and GHG emissions- and their trade-offs by using a biogeochemical model in an irrigated corn production system. The integrated assessment of AES and AEDS has been developed also from an emergy-based approach to concrete agroecosystems in China (Ma et al., 2015) and Pakistan (Shah et al., 2019), accounting the relationships among natural and semi-natural habitats surrounding cropland and agroecosystems in terms of AES and AEDS. Blanco et al. (2020) even considered farmers' attitudes and perceptions regarding the management of rural forests in agricultural landscapes in France.

The economic valuation focus can also be broadened by accounting for AEDS, which are rarely used in integrated economic valuation. As far as we know, only a few studies have addressed the economic valuation of AES and AEDS from an integrated perspective. Chang et al. (2011) estimated the net value of AES provided by greenhouse vegetable cultivation compared to conventional cultivation, in China, by employing food production, $CO_2$ fixation, soil retention and soil fertility as AES, and irrigation water use, NO3- accumulation and $N_2O$ emissions as AEDS indicators. Similarly, Hardaker et al. (2020) estimated the value of agricultural uplands in Wales, taking livestock and crop production, water supply, carbon sequestration and employment as AES flows, and water quality reduction and greenhouse gases emissions as AEDS flows. Sandhu et al. (2020), for their part, estimated the economic value associated with the benefits and costs of corn production systems in Minnesota (US) by adapting the TEEBagrifood framework (TEEB, 2018), which considers not only AES and AEDS flows, but also the stock of the social, human and natural capital produced. Nevertheless, all these authors used direct market and cost-based methods to estimate the economic values of AES and AEDS.

The range of methodology used to assess AEDS is as broad as that available for AES (Campagne et al., 2018; TEEB, 2018). Thus, the method employed will depend on the pursued aim. Since our study aims to value monetarily AES and AEDS in an integrated way, the methodology employed will require the use of economic valuation techniques. TEEB (2018) listed direct market value approaches, cost-based methods, revealed preference approaches and stated preference methods as the main methodological frameworks applied for economic valuation of AES and AEDS.

Our work applied DCEs in order to calculate the monetary value associated with the AES and AEDS provided by agriculture. This method was used because, based on microeconomic utility theories, it models trade-offs among AES and AEDS and allows measurement of the net wellbeing impact of agroecosystems. Wellbeing is, in turn, the root of the AES and AEDS concept. The employment of DCEs has grown in recent years, with many purposes, such as analysis of the economic value of water for irrigation (Rigby et al., 2010), the design of agri-environment schemes (Vaissière et al., 2018) and assessment of the demand for AES (Jourdain and Vivithkeyoonvong, 2017; Tienhaara et al., 2020).

Focusing on agroecosystems, previous studies were aimed at understanding the social preferences for specific AES, such as pollination (Breeze et al., 2015), biodiversity (Varela et al., 2018) and soil carbon sequestration (Glenk and Colombo, 2011; Rodríguez-Entrena et al., 2014), or for a set of AES provided by a specific type of agroecosystem. Several of these are now cited. Rodríguez-Ortega et al. (2016) developed an economic valuation of the AES provided by Mediterranean high nature value farmland, specifically the landscape aesthetic, biodiversity, forest fire prevention and supply of quality products. Jourdain and Vivithkeyoonvong (2017) estimated the social demand for food production, drought mitigation, water quality and the maintenance of the rural lifestyle in irrigated rice agroecosystems in Thailand. Novikova et al. (2017) broadened the scope of their assessment to include the entire country of Lithuania and valued the preferences for the reduction of underground water pollution, biodiversity and maintenance of agricultural landscapes as the main AES. Granado-Díaz et al. (2019) focused on olive groves in Andalusia (Spain) and valued the social demand for soil erosion prevention, carbon sequestration and the biodiversity provided in such agroecosystems. Notwithstanding, to our knowledge, DCEs have not been applied yet to the economic valuation of AEDS, or to the integrated valuation of AES and AEDS, which adds to the novelty of the present work.

The contribution of this paper to the on-going research into the economic valuation of AES and AEDS is two-fold. Firstly, it represents the first non-market valuation which integrates AES and AEDS in a common framework by using stated preference methods. It is expected, therefore, that the results will provide a better insight into social preferences for agriculture and the relationship between agroecosystems and human wellbeing. Secondly, we address the main agri-environmental challenges facing water-scarce Mediterranean agroecosystems. The case study of the Region of Murcia comprises a region where a great variety of agroecosystems exist -from rainfed to highly-intensive agroecosystems- and where the human and agricultural pressures threaten the surrounding ecosystems and even the inner agroecosystem functioning. Hence, we expect the present work will serve to better inform the design and implementation of current and future agricultural policies in areas with similar characteristics.

## 3.3. Material and methods

### 3.3.1. Case study

The case study is the agroecosystems of the Region of Murcia (south-eastern Spain), within the Segura River Basin (Figure 3.1). This region, bordering the Mediterranean coast, is characterised by a semiarid climate with low rainfall (< 400 mm/year) and high mean annual temperatures (between 10 and 18 °C); hence, water scarcity is one of its main characteristics. The existence of good-quality soils has fostered the development of a very important agricultural sector here. Relevant environmental challenges in the area are the soil degradation, groundwater overexploitation and salinisation and biodiversity loss. These agri-environmental characteristics make the Region of Murcia a case study representative of most semiarid Mediterranean regions (Martínez-Paz et al., 2018).

The agroecosystems within the Region of Murcia can be classified into three different sub-systems regarding their geomorphological characteristics, water availability and inputs-outputs relations with other ecosystems. There is a rainfed agroecosystem and an irrigated one, which can be further divided into a traditional irrigated agroecosystem (Heider et al., 2018) and a highly-intensive irrigated agroecosystem (Alcon et al., 2017). The rainfed agroecosystem covers around 253,000 ha (CARM, 2017), which represents 57% of the total cropland. Water scarcity determines the crop typology: almonds and olive orchards, as woody crops, and cereals, among the herbaceous crops, predominate. Irrigated agroecosystems - traditional (25%) and highly-intensive (75%) - cover 188,000 ha (CARM, 2017). The traditional irrigated agroecosystem follows the Segura River valley, citrus orchards being the main crop. It is recognisable by its landscape, well-known as the Huerta of Murcia, with high social and cultural values (Martínez-Paz et al., 2019). The highly-intensive irrigated agroecosystem occupies the lowlands, spreading from the south to the north of the Region along the Mediterranean coastline: horticultural crops and citrus are the main crops and their production is export-oriented.

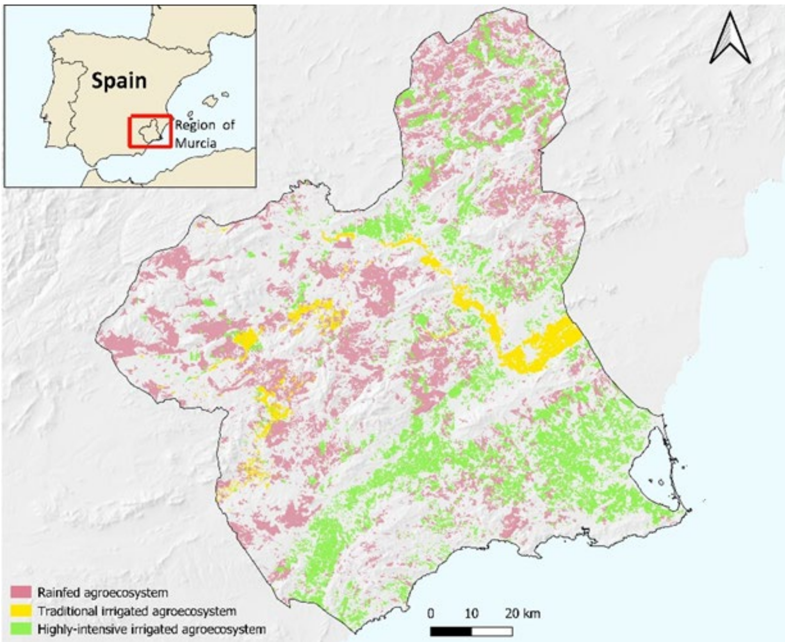


Figure 3.1. Study area.

## 3.3.2. Discrete choice experiment (DCE)

The DCE is a stated preference method based on the multi-attribute utility (Lancaster, 1966) and random utility (McFadden, 1974) theories. The fundaments of this method can be found in Champ et al. (2017).

### *3.3.2.1. Experimental design*

The implementation of a DCE requires, as a first step, the selection and definition of the attributes and their levels. Table 3.1 summarises the six attributes (and associated indicators, as well as their levels) used in the experiment. These attributes were selected based on a consultation with experts. We considered as experts those stakeholders involved in agri-environmental management in the Region of Murcia. Four different groups of stakeholders were consulted: (i) users, which included farmers, agricultural engineers and technicians, etc.; (ii) researchers from the public and private sectors involved in agricultural, ecosystem services and economic research; (iii) public managers, which encompassed agricultural and water management authorities; and (iv) civil society, mainly representatives from political parties, environmental associations and NGOs. The sample comprised 44 experts, equally distributed among the groups of stakeholders. The consultation with the experts was developed by face-to-face interviews, between July and September 2018, and a DCE was also used, after a pre-selection of 12 attributes based on a literature review (see Zabala et al., 2021). Based on the experts´ opinions, attributes were chosen to include the overall agricultural contributions to human wellbeing, but focusing on the main agri-environmental challenges of the case study. The levels of the attribute indicators deal with the three different agroecosystems found in the Region of Murcia.

Food provision constitutes the main provisioning service provided by these agroecosystems. Hence, “yield”, measured in terms of final production per unit area, was used as the indicator of this service, following Jourdain and Vivithkeyoonvong (2017). Almond production was chosen as representative since almond is the only crop present in all three agroecosystems of this case study. In addition, three levels, each associated with one agroecosystem, were assessed: rainfed (< 500 kg/ha/year), traditional irrigated (500-1,000 kg/ha/year) and highly-intensive irrigated (1,000-2,000 kg/ha/year). This indicator was used to assess the contribution of the agroecosystems to food security (Cooper et al., 2009; Villanueva et al., 2018).

The second attribute was the water supply for irrigation, which was considered among the AEDS due to the water scarcity that dominates in the study area and generates rivalry for water resources among competing uses (Zabala et al., 2019). This implies not only the need for economic and water-allocation solutions, but also social and environmental challenges (Perni and Martínez-Paz, 2017). All three agroecosystems were included within this attribute, since the rainfed agroecosystem was comprised by a zero-water supply level.

The third attribute was the local climate-regulation service, due to the impact of agriculture on the mitigation of the effects of climate change. Temperature variation on the land surface was selected as the indicator. The selected levels ranged from 0 to 2ºC of temperature reduction, corresponding to the values of the rainfed and irrigated agroecosystems, respectively (Albaladejo-García et al., 2020).

The fourth attribute was groundwater pollution. The contribution of agroecosystems to water purification and waste treatment is expected to be negative, which means it should be considered among the AEDS (Shackleton et al., 2016). Groundwater pollution is one of the main agri-environmental challenges nowadays, largely due to the salinisation of water bodies, which can be caused by nitrate discharges from agriculture (Alcolea et al., 2019). The selected levels cover the different nitrate concentrations measured in the aquifers of the case study, which can be associated with each type of agroecosystem found in the Region of Murcia (CHS, 2017). To our knowledge, although groundwater quality has also been included as an attribute in other agroecosystem valuations (Niedermayr et al., 2018; Tienhaara et al., 2020), this is the first time that it has been considered among the AEDS.

The fifth attribute was biodiversity. This attribute was measured as the bird species richness indicator. The selection of this indicator was motivated by the fact that bird richness has been reduced in the last few years due to agricultural activity (Beckmann et al., 2019), as well as by the ease with which it is understood by ordinary citizens, as proved in several other agroecosystem valuations (Varela et al., 2018). The relationship between biodiversity and agricultural intensity is not linear (Beckmann et al., 2019). Indeed, this relationship is greatly influenced by agricultural practices (Aguilera et al., 2020). Crop diversity and heterogeneous landscapes enhance bird species richness (Stjernman et al., 2019), even in fruit-tree crops (Rime et al., 2020). In the present work, biodiversity levels were defined as the share of the potential bird richness which could be found in the agroecosystems, following the Perni and Martínez-Paz (2017) procedure. Thus, low-intensity agroecosystems together with heterogeneous landscapes, such as the traditional irrigated agroecosystem in the case study, provide greater bird species richness. Therefore, it is expected that the highly-intensive irrigated agroecosystem, dominated by monoculture, would exhibit a value of 60% for bird richness with respect to the potential richness, while for the rainfed agroecosystem, of low intensity but with homogeneous landscapes, the value would be 80%.

The sixth non-monetary attribute was related to the cultural contribution of agroecosystems to human wellbeing, another of the AES. This was measured by means of a dummy indicator which reveals the chance of enjoying agroecosystems. Although most agroecosystem valuations have included the aesthetic landscape as a cultural service (e.g. Rodríguez-Ortega et al., 2016; Niedermayr et al., 2018), this experiment considers recreation, leisure and the contribution to ecotourism. These comprise the benefits derived from enjoying agricultural landscapes through participation in, for instance, sporting activities, farm tours, and bird watching, according to the experts consulted.

The monetary attribute referred to the reallocation of household taxes (Rogers et al., 2020) to support agricultural policies in the Region of Murcia; this currently amounts to around 6 €/household/month (CARM, 2018). Five levels were included, extending above and below this value and ranging from 0 to 12 €/household/month. Thus, the respondents could choose levels below or above the current value.

Table 3.1. Attributes and levels.

| | AES/AEDS | Attribute | Definition (Indicator) | Units | Levels |
|---|---|---|---|---|---|
| Provisioning services | Food provision (AES) | Yield (FOOD) | Annual almond yield produced by the agroecosystem | kg/ha/year | < 500<br>500-1,000<br>1,000-2,000 |
| | Water (AEDS) | Water supply for irrigation (WATER) | Irrigation water supplied to the crop system | $m^3$/ha/year | 0<br>< 3,000<br>3,000-5,000<br>> 5,000 |
| Regulating services | Climate regulation (AES) | Temperature regulation (TEMPE) | Temperature changes on the land surface due to agriculture | ºC | 0<br>-1ºC<br>-2ºC |
| | Water purification and waste treatment (AEDS) | Groundwater pollution (POLL) | Nitrate concentration in aquifers | mg $NO_3^-$/L | < 50<br>50-200<br>> 200 |
| | Maintenance of genetic diversity (AES) | Biodiversity (BIOD) | Bird species richness as a share with respect to potential | % | 60 %<br>80 %<br>100 % |
| Cultural services | Opportunities for recreation and tourism (AES) | Recreation and tourism (RECRE) | Chance of enjoying activities in the agroecosystems | - | No<br>Yes |
| | | Monetary payment (COST) | Part of current paid taxes directed to support agricultural policies (tax redistribution) | €/household/ month | 0<br>3<br>6<br>9<br>12 |

The attributes and levels were combined through a Bayesian efficient design, using as priors the coefficients estimated with a conditional logit model developed after the consultation with experts (see Zabala et al., 2021). The choice-sets were generated with Ngene software (ChoiceMetrics, 2012). The Bayesian D-error for the final design was 0.000194. The final design resulted in 20 choice-sets grouped in 4 blocks. Each choice-set was composed of 3 unlabelled alternatives (Jin et al., 2017), which represented different agroecosystems. The respondents were asked to choose the agroecosystem that they would like to be implemented in the Region of Murcia, according to their preferences and budget restriction. Besides, the numerical attribute levels were encoded as categorical (high, medium and low levels) to ease their understanding during the survey (Barkmann et al., 2008) and to avoid the endpoint problem (Kontogianni et al., 2010). The fifth choice-set in every block was readapted to include the assessment of preference monotonicity. For this, the third alternative in each fifth choice-set encompassed a dominated choice alternative (Mattmann et al., 2019). If an individual chooses this alternative, it reveals their non-monotonous preferences and, thus, this observation should be removed from the sample. The attribute levels within the choice-sets were presented with visual aids to make their understanding easier. An example of a choice-set is provided in Appendix 3.A.

A forced choice design was employed. This design was selected due to the aim of the research: we intended to value the AES and AEDS, not the changes in their provision because of specific agricultural policies. So, there was no need to include an opt-out option covering the non-support of a specific policy in each choice-set. Besides, since there are different types of agroecosystems within the case study, it would be inviable to define a fixed agroecosystem as a status quo or business as usual alternative. In this context, the forced choice design prevents the respondents from selecting the opt-out alternative as a strategy to elude the cognitive effort of revealing their preferences (Rigby et al., 2010; Alemu and Olsen, 2018). Nevertheless, a zero-cost level was also included in the design, to cover non-preferences in the public support of agriculture.

#### *3.3.2.2. Data collection*

Data were collected between January and February 2019, by face-to-face interviews. These were conducted by trained enumerators. An information brochure was given to the respondents to provide specific information about the definition of the attributes, indicators and levels. The target population was the households of the Region of Murcia (539,000 households), the final sample comprising 433 households. Households were randomly selected, following a stratification by county. The survey was administrated in public spaces, such as markets, parks and squares, in order to ensure the randomness of the sample. The sample size, for a 95% confidence level, provided a sample error term below 5%.

Steps were taken to mitigate hypothetical bias, following the recommendations of Loomis (2014). Specifically, two ex-ante strategies were applied: (1) the respondents were noticed that the survey results would be used to inform agricultural policies and, so, would have a consequent impact on the public budget distribution; (2) a brief cheap-talk about the aims of the research and the definitions of AES and AEDS was provided. Champ et al. (2009)

demonstrated that cheap-talk is effective when the respondents are not familiar with the goods or services to be valued, which is the case of AES and AEDS. In fact, 88% of the respondents admitted not knowing about the concepts of AES and AEDS before the interview.

### 3.3.3. Econometric and valuation framework

According to the random utility theory (McFadden, 1974), the utility ($U_{ijt}$) provided for an individual *i* from choosing an agroecosystem alternative *j* in a choice set *t* can be decomposed into an observed ($V_{ijt}$) and an unobserved part ($\varepsilon_{ijt}$), considered additively:

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt} = \sum_{k=1}^{K} \beta_{ik} X_{kjt} + \varepsilon_{ijt} \tag{3.1}$$

Where $V_{ijt}$ is the deterministic part of the utility, determined by the *k* attribute levels ($X_{kjt}$), and $\varepsilon_{ijt}$ is a stochastic error term, identically and independently distributed following a Gumbel-distribution. Assuming $V_{ijt}$ to be a weighted sum of the attribute levels, $\beta_{ik}$ is the individual marginal utility obtained from each of the *k* indicators for AES and AEDS, reflecting how the utility level changes if the provision of AES and AEDS increases.

However, despite its wide use, a linear utility function (Equation 3.1) is not always the best way to model social preferences, since marginal utility could be non-constant. Preferences for goods and services tend not to be linear, but to have a concave form. In order to deal with this, terms describing the interactions among attributes have been included in the assessment (Karaca-Mandic et al., 2012). Hence, one can distinguish squared terms of the continuous attributes, which generate a quadratic utility function, from the interactions terms among different attributes. This adds to the analysis of the relationship among the attributes considered in the interaction. Applying both types of interactions terms to the model specification gives:

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt} = \sum_{k_1=1}^{K} \beta_{ik_1 1} X_{k_1 jt} + \sum_{k_1=1}^{K} \beta_{ik_1 2} X_{k_1 jt}^2 + \sum_{k_1,k_2} \beta_{ik_1 k_2} X_{k_1 jt} X_{k_2 jt} + \varepsilon_{ijt} \tag{3.2}$$

$$\forall\, k_1, k_2 = 1, \dots, K, \qquad k_1 \neq k_2$$

The model applied most commonly to estimate the utility function is the mixed logit (MXL). It allows the coefficients to be individual-specific, through the assumption that they follow a density function *β ~ f (β|ρ)*, *ρ* being the set of parameters which describe their distribution. This permits one to model unobserved heterogeneity across individuals, and to overcome the independence of irrelevant alternatives (Hensher et al., 2005). The MXL model is estimated using the maximum simulated likelihood estimator (Train, 2009). Specifically, the utility function was modelled in R software (R Core Team, 2019), using the Apollo package (Hess and Palma, 2019) and 500 Halton draws for the simulation of the log-likelihood function.

The economic value of AES and AEDS is estimated using the marginal rate of substitution (MRS). When a cost attribute is included in the DCE, the MRS between the non-cost attributes and the cost attribute shows the willingness to pay (WTP) for the non-cost attributes. Following Equation 3.3, it is calculated as follows:

$$MRS_c^{k_1} = WTP_{k_1} = \frac{\partial U_{ijt} / \partial X_{k_1 jt}}{\partial U_{ijt} / \partial COST} = -\left(\frac{\beta_{k_1 1} + 2\beta_{k_1 2} X_{k_1} + \beta_{k_1 k_2} X_{k_2}}{\beta_c}\right) \tag{3.3}$$

Where $\beta_c$ refers to the marginal utility of the cost attribute. Since this specification could imply non-constant marginal utility for some of the attributes included, $X_{k_{1,2}}$ represents the provision level for the mentioned attributes $k_1$ and $k_2$, respectively. $WTP_{k_1}$ represents, in monetary terms, how much the respondents are willing to pay for a unit increase in each AES or AEDS $k_1$ provided by the agroecosystem.

In order to estimate how a certain provision level of AES or AEDS impacts on human wellbeing, we need to calculate the consumer surplus (CS) associated with this provision level. It can be derived as follows (Freeman et al., 2014):

$$CS_{k_1}\left(X_{0k_1}\right) = \int_0^{X_{0k_1}} WTP_{k_1}\, dX_{k_1} = \int_0^{X_{0k_1}} -\left(\frac{\beta_{k_1 1} + 2\beta_{k_1 2} X_{k_1} + \beta_{k_1 k_2} X_{k_2}}{\beta_c}\right) dX_{k_1} \tag{3.4}$$

Where $CS_{k_1}\ (X_{0k_1})$ represents the consumer surplus associated with the AES or AEDS $k_1$ evaluated at provision level $X_{0k_1}$.

Aggregating $CS_{k_1}$ for the $k_1 = 1, \dots, K$ AES and AEDS provided by an agroecosystem, the total economic value (TEV) provided by the agroecosystem can be calculated:

$$TEV = \sum_{k_1=1}^{K} CS_{k_1}\left(X_{0k_1}\right) \tag{3.5}$$

### 3.3.4. Sample characteristics

The sample comprised 433 households. Descriptive statistics for the main sociodemographic characteristics of the sample are shown in Table 3.2. The sample was totally representative of the regional census data in terms of gender, monthly income and educational level, which ensures the results represent social preferences. Furthermore, 17% of the respondents admitted that at least one household member worked in farming. This is also representative in terms of the active population, guaranteeing an appropriate distribution between farmer and non-farmer-related households.

Table 3.2. Sample and population descriptive statistics.

| Variable | | Sample | Region of Murcia | |
|---|---|---|---|---|
| *Sociodemographic information* | | | | t-test (p-value) |
| Age (years) | | 43.36 | 47.90[a] | -4.91 (0.00) |
| Gender (% women) | | 50.81 | 50.32[a] | 0.20 (0.84) |
| Household income (€/month) | | 2,406 | 2,429[b] | -0.43 (0.67) |
| Educational level (%) | | | | Pearson $\chi^2$ (p-value) |
| | Lower education | 6.70 | 10.00[c] | 1.25 (0.74) |
| | Primary education | 8.08 | 9.80[c] | |
| | Secondary education | 44.80 | 46.60[c] | |
| | Higher education | 40.42 | 33.50[c] | |
| *Relation to farming* | | | | |
| Does any member of your household work in farming? (%) | | 16.86 | 13.40[c] | 1.92 (0.06) |

[a] INE (2018a); [b] INE (2018b); [c] INE (2019)

## 3.4. Results

### 3.4.1. Estimated choice models

The social utility function was estimated employing different specifications, with a final sample of 425 observations, after removing eight cases associated with individuals who stated non-monotonous preferences (Mattmann et al., 2019). Food provision (FOOD) and water supply for irrigation (WATER) were rescaled to tonnes and $dm^3$, respectively.

Table 3.3 shows the main estimated models. Model 1 presents an *MXL-Linear* specification. The coefficient signs verify the consideration of AES and AEDS established previously. Food provision (FOOD), contribution to biodiversity (BIOD) and the chance to do recreational activities within the agroecosystem (RECRE) have a positive sign, revealing their provision has a positive impact on human wellbeing and, thus, that their consideration as AES was correctly specified. In contrast, water supply for irrigation (WATER) and groundwater pollution (POLL), which were predefined as AEDS, show negative signs. This specification implies that the marginal utility is constant and, thus, independent from the provision level of AES and AEDS.

However, microeconomics suggests the existence of concave utility functions with diminishing marginal utility. To overcome this challenge, a non-linear relationship between attributes and social utility was tested using squared attributes (Model 2) and also including interaction terms (Model 3). A step-wise procedure was followed to select the squared and interaction terms that better fitted both models; concretely, all feasible squared and interaction terms were tested (saturated models) and non-significant terms were deleted until reduced models which better explained the choices were obtained. All models were estimated assuming a normal distribution for non-monetary and squared attributes, whilst the cost coefficient and interaction terms between different attributes were set as fixed.

Model 2 shows an *MXL-Quadratic* specification. Significant coefficients of squared attributes were obtained for FOOD, WATER and POLL, revealing the non-linearity in the utility function. The coefficients of the squared attributes show negative signs, indicating the concavity of the utility function and the diminishing marginal utility provided. The *MXL-Quadratic* specification provided a better fit than the *MXL-Linear* model (LR = 106.25; $\chi^2_{0.05;6}$ = 12.59). The inclusion of interaction terms between attributes (Model 3) also improved upon Model 2 (LR = 13.53; $\chi^2_{0.05;2}$ = 5.99). Therefore, Model 3 is the preferred model to be used in the follow-up assessment. This model is an MXL model with two significant interactions: *FOOD*WATER* and *WATER*POLLUTION* (Table 3.3).

Table 3.3. Estimation results from MXL models.

| | Model 1 *MXL - Linear* | | | Model 2 *MXL - Quadratic* | | | Model 3 *MXL – All interactions* | | |
|---|---|---|---|---|---|---|---|---|---|
| | *β* | SE | | *β* | SE | | *β* | SE | |
| *Mean* | | | | | | | | | |
| FOOD | 0.95 | 0.07 | *** | 4.14 | 0.68 | *** | 4.58 | 0.67 | *** |
| WATER | -0.13 | 0.02 | *** | 0.62 | 0.12 | *** | 0.73 | 0.12 | *** |
| TEMPE | 0.00 | 0.04 | | 0.08 | 0.05 | * | 0.07 | 0.04 | * |
| POLL | -0.01 | $5.43 \cdot 10^{-4}$ | *** | $-3.84 \cdot 10^{-3}$ | $1.74 \cdot 10^{-3}$ | ** | $-3.76 \cdot 10^{-3}$ | $2.06 \cdot 10^{-3}$ | * |
| BIOD | 0.01 | $2.23 \cdot 10^{-3}$ | ** | 0.01 | $2.65 \cdot 10^{-3}$ | *** | 0.02 | $2.68 \cdot 10^{-3}$ | *** |
| RECRE | 0.45 | 0.08 | *** | 0.59 | 0.09 | *** | 0.67 | 0.10 | *** |
| COST | -0.04 | 0.01 | *** | -0.06 | 0.01 | *** | -0.07 | 0.01 | *** |
| $FOOD^2$ | | | | -1.29 | 0.26 | *** | -1.33 | 0.25 | *** |
| $WATER^2$ | | | | -0.12 | 0.02 | *** | -0.11 | 0.02 | *** |
| $POLL^2$ | | | | $-1.34 \cdot 10^{-5}$ | $4.90 \cdot 10^{-6}$ | *** | $-9.26 \cdot 10^{-6}$ | $4.86 \cdot 10^{-6}$ | * |
| FOOD*WATER | | | | | | | -0.10 | 0.03 | *** |
| WATER*POLL | | | | | | | $-3.82 \cdot 10^{-4}$ | $1.75 \cdot 10^{-4}$ | ** |
| *SD* | | | | | | | | | |
| FOOD | 0.64 | 0.09 | *** | 0.02 | 0.39 | | -0.45 | 0.35 | |
| WATER | -0.13 | 0.06 | ** | -0.03 | 0.11 | | 0.02 | 0.03 | |
| TEMPE | -0.35 | 0.10 | *** | -0.27 | 0.13 | ** | -0.29 | 0.12 | *** |
| POLL | $4.73 \cdot 10^{-3}$ | $5.99 \cdot 10^{-4}$ | *** | $3.73 \cdot 10^{-3}$ | $1.45 \cdot 10^{-3}$ | *** | $4.23 \cdot 10^{-3}$ | $8.95 \cdot 10^{-4}$ | *** |
| BIOD | 0.02 | $4.58 \cdot 10^{-3}$ | *** | -0.02 | 0.01 | *** | -0.02 | 0.01 | *** |
| RECRE | 0.76 | 0.13 | *** | -0.79 | 0.13 | *** | 0.80 | 0.14 | *** |
| $FOOD^2$ | | | | -0.30 | 0.04 | *** | -0.24 | 0.10 | *** |
| $WATER^2$ | | | | -0.03 | 0.01 | *** | 0.03 | 0.01 | *** |
| $POLL^2$ | | | | $1.29 \cdot 10^{-3}$ | $3.02 \cdot 10^{-6}$ | *** | $9.19 \cdot 10^{-6}$ | $2.53 \cdot 10^{-6}$ | *** |
| LL | | -1,785.37 | | | -1,732.25 | | | -1,725.48 | |
| $R^2$-Adjusted | | 0.23 | | | 0.25 | | | 0.25 | |
| AIC | | 3,596.75 | | | 3,502.50 | | | 3,492.97 | |
| BIC | | 3,670.35 | | | 3,610.06 | | | 3,611.86 | |

Statistically significant at a level of *0.1, **0.05, and ***0.01.

For Model 3, the mean coefficients are significant at least at the 10% level, while the standard deviation estimations are significant for all attributes except FOOD and WATER. These results reveal that the perceived impact of food provision and water supply for irrigation on human wellbeing was homogenous across the respondents.

The mean coefficients for AES and AEDS have the expected sign. Food provision has a positive sign, which reveals that people feel human wellbeing when agriculture provides society with food. However, it shows diminishing marginal utility, as revealed by the negative sign of the squared coefficient. This implies that a high level of food production provides decreasing levels of marginal utility; that is, increments in food production are expected to have higher positive effects on utility when production is low.

Similar statements could be applied to the case of water supply for irrigation. In relation to this attribute, people are aware of the importance of using water for agriculture, and they even consider that they get utility when some water is supplied to the agricultural sector. However, it should not be done on an unlimited basis. The negative sign of $WATER^2$ shows diminishing marginal utility, revealing that alternative uses for the water destined to irrigation could be preferred by the respondents under some circumstances.

The interaction between FOOD and WATER is also significant, revealing the negative relationship between them. The utility provided by food provision depends on the level of water supply for irrigation, and *vice versa*. Hence, high levels of water supply for irrigation reduce the marginal utility of food provision. As the water supplied for irrigation increases, there is a decline in the utility provided by food provision.

The temperature regulation (TEMPE) coefficient has a positive sign, which evidences that people also demand the cooling effect provided by agriculture, which could reach a 2ºC reduction in the case of the irrigated agroecosystems.

As expected, groundwater pollution (POLL) has a negative impact and shows diminishing marginal utility, this decrement being quicker as pollution increases. The estimated utility function also shows the significance of the interaction between WATER and POLL. High levels of groundwater pollution have negative effects on the marginal utility of the water supply for irrigation, and *vice versa*.

The respondents considered that agriculture provides an enjoyable environment that promotes recreational activities and tourism, as shown by the significant mean and standard deviation coefficients for the RECRE variable. Similarly, the agricultural contribution to biodiversity was also positively valued. The greater the bird richness in an agroecosystem, the more utility people get.

The cost coefficient has the expected negative sign, which shows the disutility people get when tax payments increase and thus provides consistency to the results.

### 3.4.2. Valuation of AES and AEDS

Table 3.4 shows the marginal WTP for AES and AEDS calculated with Model 3. The results indicate that, on average, people are willing to pay around 0.23 €/household/year in order to increase food provision. However, this WTP depends on the amount of food provided and the water supplied to achieve this level of food production. Therefore, low levels of food production and irrigation water supply will have positive and greater values of marginal WTP.

Regarding the water supply for irrigation, the results reveal that people are willing to pay for it, but they prefer that not all the available water is used for agricultural purposes. Actually, a negative WTP shows that people are willing to pay to reduce the water supply to irrigation to the level which maximises their utility: the satiation point, around 2,600 $m^3$. For instance, if the water supplied to agriculture is around 2,000 $m^3$/ha/year, people are willing to pay 0.02 €/$m^3$ to increase its availability for agriculture. However, if agriculture actually uses around 4,000 $m^3$/ha/year, people are willing to pay 0.06 €/$m^3$ to reduce the water supplied for irrigation and promote alternatives uses. Furthermore, the marginal utility of WATER also depends on the level of FOOD –the higher the food provision, the less people are willing to pay to increase the water supply for irrigation– and the level of groundwater pollution –the higher the groundwater pollution, the less people are willing to pay to enhance the water supply for irrigation–.

Table 3.4. Marginal WTP for AES and AEDS (€/household/year).

| | Mean | Confidence interval (95%)[4] | |
|---|---|---|---|
| FOOD (€/kg/ha)[1] | 0.23 | 0.18 | 0.33 |
| WATER (€/$m^3$/ha)[2] | -0.04 | -0.05 | -0.03 |
| TEMPE (€/°C) | 13.21 | -3.74 | 30.74 |
| POLL (€/mg $NO_3^-$/L)[3] | -1.42 | -1.91 | -1.09 |
| BIOD (€/p.p.) | 2.67 | 1.69 | 4.20 |
| RECRE | 117.64 | 78.33 | 177.39 |

[1] The marginal WTP of FOOD was evaluated at the mean levels of the attributes FOOD (1,100 kg/ha) and WATER (3,454 $m^3$/ha)
[2] The marginal WTP of WATER was evaluated at the mean levels of the attributes FOOD (1,100 kg/ha), WATER (3,454 $m^3$/ha) and POLL (158 mg $NO_3^-$/L)
[3] The marginal WTP of POLL was evaluated at the mean levels of the attributes WATER (3,454 $m^3$/ha) and POLL (158 mg $NO_3^-$/L)
[4] Obtained using bootstrapping (1000 samples)

The results contribute to the consideration of groundwater pollution as one of the AEDS. The WTP for this attribute is negative across all the levels considered. This has two direct implications: (1) the desired level of pollution is zero; and (2) people are willing to pay in order to reduce groundwater pollution. Moreover, this attribute shows diminishing marginal utility, revealing that the higher the pollution, the more people are willing to pay to reduce it. For instance, if groundwater pollution reaches the mean level (158 mg $NO_3^-$/L), people are willing to pay around 1.42 €/mg $NO_3^-$ to reduce it.

People are willing to pay, on average, 13.21 €/year/household in order to support policies which contribute to a 1ºC temperature reduction. The WTP for supporting agricultural policies which contribute to biodiversity is, on average, 2.67 €/year/household per percentage point (p.p.) increment. Recreational and leisure activities within agricultural landscapes also contribute to wellbeing, and are valued at 117.64 €/year/household, on average.

The results of the WTP analysis allowed estimation of the value of agroecosystems with different provision levels of AES and AEDS. For this, the provision levels for the three most representative agroecosystems of the case study were obtained from Alcon et al. (2013) and Almagro et al. (2016) in the case of food production and water supply for irrigation of almond orchards, Albaladejo-García et al. (2020) in the case of temperature and following Perni and Martínez-Paz (2017) in the case of biodiversity. Groundwater pollution levels were obtained from the water authority in charge of the regional water management (CHS, 2017).

Table 3.5 summarises the decomposition of the global value obtained for each agroecosystem. The traditional irrigated agroecosystem was the most valued agroecosystem, with a TEV of about 988 €/household/year, followed by the rainfed agroecosystem (with a TEV of around 667 €/household/year) and the highly-intensive irrigated agroecosystem, which provides 500 €/household/year of utility to the people of the Region of Murcia.

Table 3.5. AES and AEDS levels, CS and TEV. Valuation of agroecosystems (€/household/year).

| | Rainfed agroecosystem | | Traditional irrigated agroecosystem | | Highly-intensive irrigated agroecosystem | |
|---|---|---|---|---|---|---|
| | Level | Value | Level | Value | Level | Value |
| FOOD | 500 kg | 345.31 | 1,000 kg | 538.98 | 2,000 kg | 538.96 |
| WATER | 0 m³ | 0.00 | 2,000 m³ | 126.85 | 4,000 m³ | -5.36 |
| TEMPE | 0 ºC | 0.00 | -1 ºC | 13.21 | -2 ºC | 26.42 |
| POLL | 25 mg $NO_3^-$/L | -9.38 | 125 mg $NO_3^-$/L | -75.75 | 250 mg $NO_3^-$/L | -219.47 |
| BIOD | 80% | 213.86 | 100% | 267.32 | 60% | 160.39 |
| RECRE | Yes | 117.64 | Yes | 117.64 | No | 0.00 |
| TEV | | 667.43 | | 988.25 | | 500.95 |

Aggregating these values across the target population (539,000 households), the TEV could be calculated for each agroecosystem, as well as for the entire case study (Table 3.6). Thus, the whole agroecosystem provides more than 350 M€/year of human wellbeing, equivalent to 794 €/ha/year, which represents around 22% of the agricultural gross value added of the case study.

Table 3.6. TEV extension. Valuation of the agroecosystems in the Region of Murcia.

| | Rainfed agroecosystem | Traditional irrigated agroecosystem | Highly-intensive irrigated agroecosystem | Total Region of Murcia |
|---|---|---|---|---|
| Area (ha) | 253,269 | 48,077 | 139,757 | 441,103 |
| TEV (€/year) | 206,555,381 | 58,056,779 | 85,549,348 | 350,161,508 |

## 3.5. Discussion

### 3.5.1. Looking into the results

The aim of this work was to integrally value both AES and AEDS in order to integrate them into a common framework for agroecosystem valuation. Regarding the economic value of the AES and AEDS, the marginal WTP for food production, which could reach a maximum of 0.81[2] €/kg according to our results, is less than the market price received by almond farmers, which averages 1.32 €/kg (CARM, 2019). This reveals that people are not willing to support private benefits from agriculture (Jourdain and Vivithkeyoonvong, 2017), but they do value the contribution of agroecosystems to food security. Martínez-Paz et al. (2019) also found that, of the AES, fruit and vegetable production was the one valued most in the Huerta of the Region of Murcia. These authors showed that people living near the city of Murcia valued the contribution of this traditional agroecosystem to food provision at 6.83 €/household/year. This value contrasts with the results obtained in this study for the traditional irrigated agroecosystem (538.98 €/household/year), since our results consider the agricultural contribution to food security.

The supply of water for irrigation is socially supported, and the WTP can reach 0.12[3] €/m$^3$. This value represents, on average, one-third of the current price paid by farmers (CCRC, 2019). These results imply people are willing to support the use of water for irrigation. However, the diminishing marginal utility means that this WTP will depend on the current level of water supply for irrigation, as well as the level of food provision and groundwater pollution. It also means that this economic value could become negative, which indicates that the use of additional water for irrigation would be translated into a social cost. The satiation point for WATER, around 2,600 m$^3$, establishes the boundary between positive wellbeing and social cost. Rigby et al. (2010) also estimated the value of irrigation water for farmers in the Region of Murcia using a DCE and obtained a mean WTP of 0.45 €/m$^3$. Therefore, it seems the private value of irrigation water is higher than its public value.

The value of agricultural services regarding climate regulation has been estimated in most cases according to the social demand for reduction of $CO_2$ emissions, or the improvement in $CO_2$ sequestration due to agricultural activity (Granado-Díaz et al., 2019). However, people do not perceive these flows as an agricultural impact on their wellbeing, but they do perceive

[2] Marginal WTP of FOOD evaluated at the zero level of the attributes FOOD (0 kg/ha) and WATER (0 m$^3$/ha)
[3] Marginal WTP of WATER evaluated at the zero level of the attributes FOOD (0 kg/ha), WATER (0 m$^3$/ha) and POLL (0 mg $NO_3^-$/L).

changes in the local temperature as an agricultural effect on climate regulation. In fact, we estimate that people are willing to pay around 13 €/household/year per degree of temperature reduction.

The literature concerning the non-market value of groundwater pollution is scarce, but there are many reports related to the economic valuation of water quality in agricultural contexts. Niedermayr et al. (2018) estimated that the value of groundwater which is able to be used without treatment ranges between 66 and 87 €/household/year, in an area in the northeast of Austria. Similarly, Jourdain and Vivithkeyoonvong (2017) estimated a value of 64 €/household/year in the case of its use for swimming purposes. These values are in line with the ones obtained in this work.

Contrastingly, biodiversity is one of the AES most employed for agroecosystem valuation using DCEs (e.g. Vaissière et al., 2018; Granado-Díaz et al., 2019). We found a WTP of 2.63 €/household/year per p.p. increment in bird diversity, very close to the values of Rodríguez-Ortega et al. (2016), who estimated that the value associated with the presence of bearded vultures in mountain agroecosystems in northeast Spain ranges between 1.82 and 2.08 €/household/year per percentage point of increment. However, our value contrasts with the one obtained by Perni and Martinez-Paz (2017) for a human-created wetland located close to the case study site: 0.18 €/household/year per percentage point of increment. These differences could be due to the extent of the two ecosystems: our work focused on all agroecosystems within the Region of Murcia, while that of Perni and Martinez-Paz (2017) was centred on one specific wetland ecosystem.

Finally, the enjoyment of leisure and recreational activities within agricultural landscapes is valued at about 110 €/household/year according to our results. However, comparison with the values obtained in other works located near to the study area, 2.85 and 2.81 €/household/year for García-Llorente et al. (2012) and Martínez-Paz et al. (2019), respectively, suggests our value is an overestimation. These differences could be related to the fact that our work focused on different agroecosystems, which include activities such as wine tourism (Cebrián and Rocamora, 2017), ecotourism and environmental education (Robledano et al., 2018), together with sport activities. Nevertheless, this result reveals that agricultural management should also include culture-friendly approaches, to integrate all dimensions of human wellbeing.

The results presented here encompass the non-market valuation of the AES and AEDS provided by the agroecosystems studied. However, their benefits and costs for society could broaden beyond the scope considered in this study. The market valuation of trading agricultural outcomes could also be integrated with the non-market values estimated here. Therefore, of the AES, food provision is the one which provides both market and non-market values to society. The market value of food provision could be summarised by the gross margin, as an indicator. For instance, assuming the almond gross margins for the rainfed, traditional and highly-intensive irrigated agroecosystems to be around, respectively, 350, 1,000 and 1,500 €/ha/year (Alcon et al., 2013; Lehtonen et al., 2020), the integrated market and non-market value of each agroecosystem rises to, approximately, 1,150, 2,200 and 2,100 €/ha/year, respectively.

Consideration of the market values of AES provides an additional perspective on the integrated contributions of agroecosystems. In fact, these values reinforce the results showing that differences in productivity cannot overcome differences in the values of AES and AEDS. The market and non-market values are lowest for the rainfed agroecosystem; however, the two irrigated agroecosystems have similar values. This reveals that greater provision of AES -and lower provision of AEDS- by traditional irrigated agroecosystems compensates differences in productivity with respect to highly-intensive irrigated agroecosystems. The integrated market and non-market values provided by both irrigated agroecosystems show that similar values could be reached with greater AES and lower AEDS. Hence, this illustrates again that higher food production is not always socially desired, but it must be considered in the overall contributions to human wellbeing.

### 3.5.2. Policy implications: Because AEDS matter

The production of enough healthy food for a growing population, while mitigating negative impacts on ecosystems and human wellbeing, is the main agricultural challenge for the next decade (Sandhu et al., 2019). This implies the integration of multiple contributions, both positive and negative, of agriculture to human wellbeing. Hence, the results of this study provide evidence that AEDS should be valued integrally together with AES. As Shackleton et al. (2016) pointed out, not considering AEDS when valuing agroecosystems may produce an overvaluation. For instance, for our results (Table 3.7), this overvaluation could reach 44% of the TEV of the highly-intensive irrigated agroecosystem.

Table 3.7. Relative importance of AES and AEDS in the TEV of different agroecosystems.

| | Rainfed agroecosystem | Traditional irrigated agroecosystem | Highly-intensive irrigated agroecosystem |
|---|---|---|---|
| FOOD | 0.52 | 0.55 | 1.08 |
| WATER | - | 0.13 | -0.01 |
| TEMPE | - | 0.01 | 0.05 |
| POLL | -0.01 | -0.08 | -0.44 |
| RECRE | 0.18 | 0.12 | - |
| BIOD | 0.32 | 0.27 | 0.32 |
| TEV | 1.00 | 1.00 | 1.00 |

At this point, the results of the present work may serve as a decision-support tool for agricultural policy makers, to improve the design and implementation of agri-environmental policies, either *ex-ante* or *ex-post*, in Mediterranean regions with water-scarcity issues. Since the results highlight the main agricultural contributions to human wellbeing and their intensity, they could be used to define policies and measures which promote these positive contributions and reduce the negative ones. In addition, this framework also allows measurement of the *ex-post* impact of agricultural policies on human wellbeing. A simple simulation exercise allows estimation of the economic values of greening actions described in the last CAP reform. These are expected to increase biodiversity (15%) and to reduce groundwater pollution (25%) (due to the reduction in fertiliser needs). Thus, based on the current situation, the impact of these measures is expected to range from 23.51 €/ha/year in the case of the traditional irrigated agroecosystem to 103.01 €/ha/year for the highly-intensive irrigated agroecosystem.

Thus, the positive impact of the greening practices will depend on the agroecosystem considered, which reveals that the efficiency of different agricultural policies would be higher if they were directed to the right agroecosystem. If only the criteria for the gain in wellbeing are considered, agri-environmental policies may focus on more degraded agroecosystems, where the expected impact is higher. However, this may imply the allocation of economic resources to those agroecosystems which pollute more, instead of rewarding the ones which actually perform better. This illustrates the challenge that arises in the design of agri-environmental policies, regarding not only socio-economic but also ethical issues, which requires the consideration of multidisciplinary and intertemporal approaches (Varela et al., 2018).

### 3.5.3. Theoretical implications: diminishing marginal utility, social demand and the interdependence of AES/AEDS

The results make it clear that attributes may not be as independent from each other as we may think. In theory, in the design of a DCE, it is considered that all attributes are independent (Hensher et al., 2005). However, this assumption may not be realistic when applied to the case of AES and AEDS. In fact, the perceived marginal utility of food provision or groundwater pollution also depends on the water supply for irrigation, and *vice versa*. Figure 3.2 summarises, *ceteris paribus*, the contributions to the total utility function of the AES and AEDS whose indicators are continuous. It also reveals this dependence among attributes.

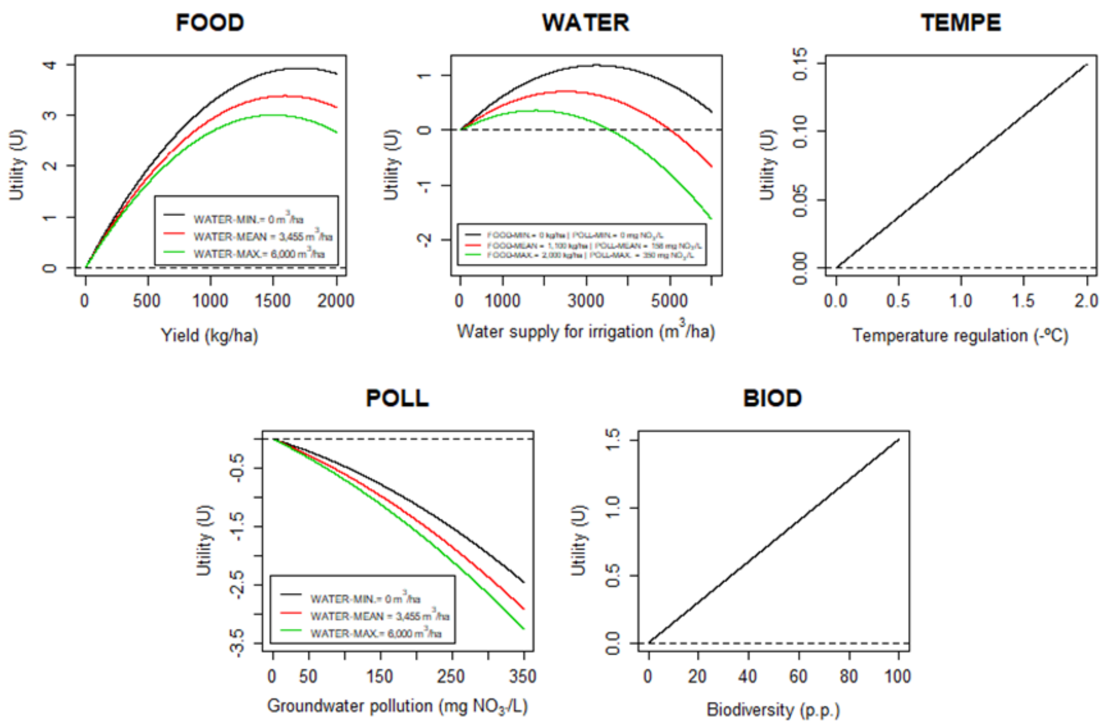


Figure 3.2. Utility functions of AES and AEDS.

Note: The utility provided by each of the AES and AEDS is presented. In particular, the utility provided by food provision (FOOD), water supply for irrigation (WATER) and groundwater pollution (POLL) depends on the value taken by some other attributes. This interdependence is shown in their respective graphs by the black (minimum expected level of the interdependent attributes), red (mean expected level of the interdependent attributes) and green (maximum expected level of the interdependent attributes) lines. The utility provided by temperature regulation (TEMPE) and biodiversity (BIO) does not depend on other attributes.

The impact of each of the AES and AEDS on human wellbeing is not linear, and also depends on the level of provision of other AES or AEDS. Food provision was expected to have a linear, positive impact on wellbeing; that is, the more food agriculture produces, the more wellbeing people get. Nevertheless, the results reveal that people prefer agroecosystems that provide food up to the satiation point (1,600 kg/ha). Translating this into the ecosystem service approach, it shows that food provision will be considered as one of the AES while the level of food provision is below this maximum - that is, while increasing the level of food provision generates positive marginal utility. This is clearly defined by the concave form of the total utility function for *FOOD* (Figure 3.2).

High levels of food production are usually linked to high levels of water supply for irrigation, and this is socially perceived as well. At this point, it is necessary to differentiate between *interaction* and *confusing* effects (Karaca-Mandic et al., 2012). The decline in the contribution of food provision to the total utility after its satiation point is related to this *confusing* effect, since high levels of food production are *confused* with high levels of water supply for irrigation and this induces the decrement in utility (see Appendix B). These results highlight that maximisation of food provision alone should not be the main focus of agricultural policy.

Similar statements could be applied to the case of water supply for irrigation. Supplying water to agriculture will provide wellbeing until its satiation point is reached. However, if water is supplied to agriculture at a level higher than this maximum, it will be considered as one of the AEDS. This is linked to the water scarcity in the case study area and reveals that, alternatively, water may be supplied to other ecosystems rather than agroecosystems. A similar situation was exposed by Zabala et al. (2019) for the competitive use of reclaimed water in agricultural irrigation or for environmental purposes. The utility provided by supplying water for irrigation is related to the level of food provided by such agroecosystems and the level of groundwater pollution. Thus, the higher the food provision and groundwater pollution, the lower the utility that people get from supplying water for irrigation (Figure 3.2), which reveals the social trade-offs among these AES and AEDS.

The agricultural contribution to groundwater pollution shows diminishing marginal utility. However, in this case, the utility is always negative, independently of the pollution level. This shows that groundwater pollution due to agricultural activity is always considered as one of the AEDS; thus, the socially-demanded level of groundwater pollution is zero. As the interaction term between *WATER* and *POLL* shows, the disutility obtained from pollution will be higher when it is perceived jointly with the water supply for irrigation. Nutrient leaching from irrigation water to groundwater is responsible for the poor ecological status of several water bodies in the case study area (Pellicer-Martínez and Martínez-Paz, 2016). Hence, there is a societal awareness, reflected in the social demand, of the physical relationship that may arise between irrigation water and groundwater pollution.

The temperature regulation and biodiversity are considered as AES, since the results show a linear, positive relationship between provision and utility.

As these results reveal, agricultural outputs can switch from AES to AEDS depending on their provision level. This idea was first presented by Rasmussen et al. (2017), and was applied to agroecosystems in Laos. A further step forward will be achieved here with the new categorisation of AES and AEDS that we propose. Thus, three main categories of AES/AEDS are suggested: (1) pure AES, for which the more that is provided, the more utility people get; (2) pure AEDS, for which the more that is provided, the more disutility people obtain; (3) quasi-AES, whose positive or negative impact on human wellbeing depends on their provision level. With this categorisation, food provision, temperature reduction and contributions to biodiversity and leisure and recreational activities can be considered as pure AES. By contrast, groundwater pollution can be placed in the pure AEDS category while, between the two extremes, water supply for irrigation may be placed in the quasi-AES category. This also evidences that AES and AEDS are not static concepts, but are context-dependent (Shackleton et al., 2016).

The theoretical implications of these results have been extrapolated to the case of marginal WTP functions (Figure 3.3) and values. Thus, pure AES are related to positive and constant, or even rising, WTP values, while pure AEDS imply negative and constant, or decreasing, values. Quasi-AES have decreasing WTP functions, which could have positive and negative sections.

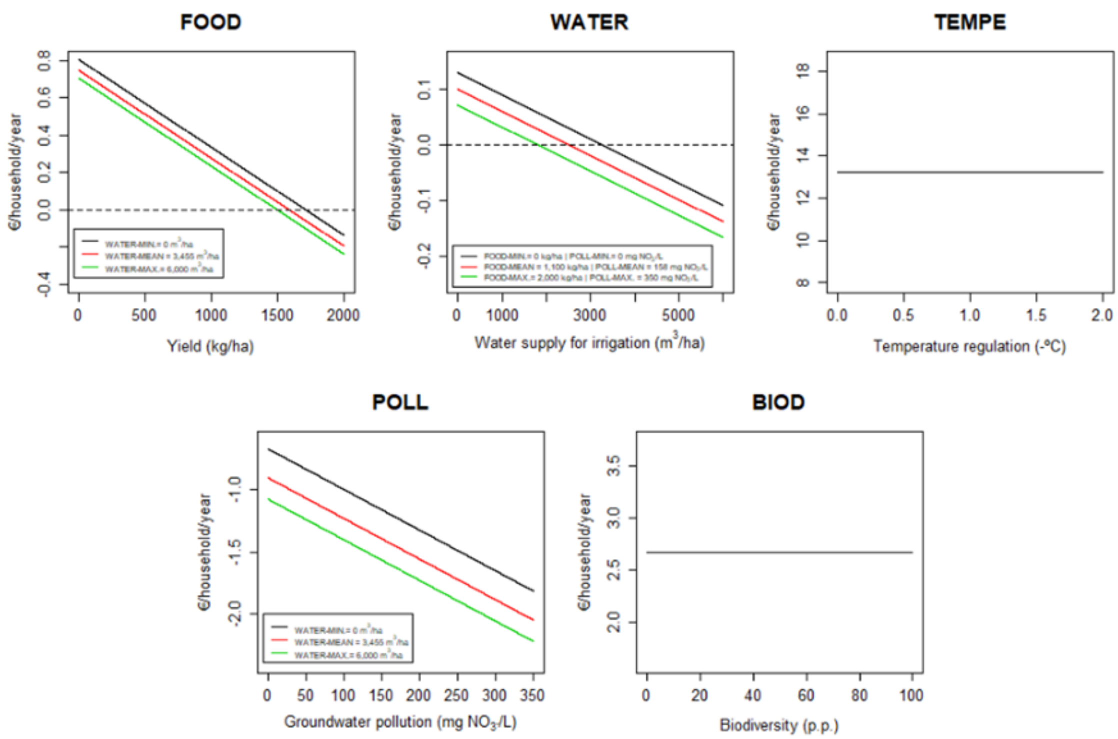


Figure 3.3. Social demand. Marginal WTP functions of AES and AEDS.

Note: The marginal WTP for each of the AES and AEDS is presented. In particular, the WTP for food provision (FOOD), water supply for irrigation (WATER) and groundwater pollution (POLL) depends on the value taken by some other attributes. This interdependence is shown in their respective graphs by the black (minimum expected level of the interdependent attributes), red (mean expected level of the interdependent attributes) and green (maximum expected level of the interdependent attributes) lines. The marginal WTP for temperature regulation (TEMPE) and biodiversity (BIO) does not depend on other attributes.

## 3.6. Conclusions

The contributions of agroecosystems to human wellbeing have been addressed here. In the main, previous studies of agroecosystems and their outputs have been focused only on the positive contributions of agriculture to human wellbeing. Hence, little is known about the negative impacts, in neither biophysical nor social terms. This study addresses the need for an integrated framework which gathers together both positive and negative agricultural outputs:

namely, AES and AEDS, respectively. For this purpose, an integrated economic valuation of the AES and AEDS provided by the agroecosystems of the Region of Murcia (south-eastern Spain) has been developed. A DCE has been employed to reach the pursued aim, using food provision, climate regulation, recreational and leisure activities and biodiversity as AES, and water supply for irrigation and groundwater pollution as AEDS.

The results show that people value both AES and AEDS, which provides a net economic valuation of the overall impact of agriculture on human wellbeing. As such, the people surveyed showed non-linear preferences for food provision, water supply for irrigation and groundwater pollution, which disclose diminishing marginal utility for these AES and AEDS. This finding also suggests that the marginal value (WTP) of these AES and AEDS depends on their provision level. Thus, social demand functions for their provision could be estimated, to calculate the value not only of each of the AES and AEDS, but also of the entire agroecosystems in the case study. Therefore, this work presents a novel framework for measuring the overall value of agriculture to society, assessing all contributions to human wellbeing.

These results will be very useful for policy makers in the development of sustainable and cost-effective agricultural measures. New agricultural policies need to deal with the environmental impacts of agricultural activity without overlooking food production and the consumption of natural resources. This could be translated into new, socially-supported agricultural policies, with agricultural practices that promote water saving, pollution mitigation, biodiversity and climate regulation. Thus, further studies may analyse the provision of AES and AEDS from a supply point of view (farmers), to explore both trade-offs and economic value and integrate them with the current assessment framework.

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## Appendix 3.A. Figures

| | Alternative A | Alternative B | Alternative C |
|---|---|---|---|
| **Almond production (kg/ha/year)** | 500-1,000 kg<br>Medium | < 500 kg<br>Low | 1,000-2,000 kg<br>High |
| **Water supply for irrigation ($m^3$/ha/year)** | > 5,000 m3<br>High | 3,000-5,000 m3<br>Medium | > 5,000 m3<br>High |
| **Temperature regulation (ºC)** | 2ºC | 1 ºC | |
| **Groundwater Pollution (mg $NO_3^-$/L)** | > 200 mg<br>High | < 50 mg<br>Low | 50-200 mg<br>Medium |
| **Recreation and tourism** | Yes | No | Yes |
| **Biodiversity (%)** | 100 %<br>High | 100 %<br>High | 60 %<br>Low |
| **Cost (€/household/month)** | **6 €** | **12 €** | **0 €** |
| **Choice** | ☐ | ☐ | ☐ |

Figure 3.A.1. Example of a choice set.

Each respondent answered one block (five choice sets), choosing only one alternative per choice-set. Each alternative represented a possible agroecosystem defined by a set of AES and AEDS, while monetary attribute denoted taxes that should be reallocated in order to support the implementation of the agroecosystem selected. Thus, each respondent should choose the most preferred agroecosystem, according to his/her preferences and budget constraint.

# Appendix 3.B. Interaction terms in utility functions from logit models

This appendix provides the mathematical implications of using interaction terms in non-linear models, analysing specifically the use of logit models to estimate utility functions and using the results from the manuscript as empirical evidence to support the theory. Following Karaca-Mandic et al. (2012), the consequences of using interaction terms, both squared and interaction terms between variables, are discussed in non-linear models. We focus on the interrelations among the variables included in the model, which needs to analyse marginal and cross-partial effects. All the expressions and formulations are developed for logit models.

## 3.B.1. Theoretical background

### *3.B.1.1. Model without interaction terms*

*Model definition*

According to the multi-attribute utility theory (Lancaster, 1966), the utility ($U$) provided by a good or service is the sum of the utility obtained by each attribute ($x_i$) that composes it. Additionally, following the random utility theory (McFadden, 1974), this utility ($U$) could be decomposed into a deterministic ($V$) and a stochastic part ($\varepsilon$), which is iid. Given the deterministic part of utility $V$, which can be dependent on two independent and continuous attributes $x_1, x_2$, and their squares, the utility ($U$) can be written as follows:

$$U(x_1, x_2) = V(x_1, x_2) + \varepsilon = \beta_{11}x_1 + \beta_{12}x_1^2 + \beta_{21}x_2 + \beta_{22}x_2^2 + \varepsilon \quad (3.B.1)$$

However, neither this linear utility function, nor its deterministic part, can be observed directly, but it can be estimated indirectly from observed choices among different alternatives of goods or services. The coefficients $\beta_k$, which maximise the probability of the observed choices, can be estimated. This implies the employment of non-linear models, such as logit models. Let $F$ be the logistic expression used to estimate indirectly the linear deterministic utility function $V$:

$$F = \frac{1}{1+e^{-V}} = \frac{1}{1+e^{-(\beta_{11}x_1+\beta_{12}x_1^2+\beta_{21}x_2+\beta_{22}x_2^2)}} \quad (3.B.2)$$

*Marginal effects*

Since the focus is to understand how the expected value of utility changes for an infinitesimal change in the value of continuous explanatory variables, the marginal utility should be obtained. Using Equation 3.B.1, the marginal utility is just the first derivative of the utility ($U$) with respect to each continuous explanatory variable $x_1$:

$$\frac{\partial U}{\partial x_1} = \frac{\partial V}{\partial x_1} = \beta_{11} + 2\beta_{12}x_1 \quad (3.B.3)$$

The marginal utility provided by attribute $x_1$ depends only on itself. Nevertheless, the utility function cannot be estimated directly, and a logistic expression needs to be used (Equation 3.B.2). Therefore, marginal effects also should take into account the non-linear model employed. Marginal effects should be obtained from the first derivative of this logit model, since it includes all the available information contained in the model estimation:

$$\frac{\partial F}{\partial x_1} = \frac{\partial F}{\partial V}\frac{\partial V}{\partial x_1} = \frac{e^V}{(1+e^V)^2}(\beta_{11} + 2\beta_{12}x_1) = \frac{e^{\beta_{11}x_1+\beta_{12}x_1^2+\beta_{21}x_2+\beta_{22}x_2^2}}{\left(1+e^{\beta_{11}x_1+\beta_{12}x_1^2+\beta_{21}x_2+\beta_{22}x_2^2}\right)^2}(\beta_{11} + 2\beta_{12}x_1) \quad (3.B.4)$$

As Equation 3.B.4 shows, when the overall estimated logit model is considered, the marginal effects of one attribute (or variable) depend on the values of the rest of the attributes. This information is also included in the estimation of the linear utility function; therefore, the marginal utility of attribute $x_1$ depends implicitly on the expected value taken by attribute $x_2$.

*Cross-partial effects*

After recognising that the marginal effects of one attribute depend also on the values taken by the rest of the attributes, even when interaction terms are not included in the model, the challenge now is to determine how this marginal utility changes for an infinitesimal change in the value of each of the other variables. To do so, cross-partial effects are obtained.

The cross-partial derivate of the utility function with respect to $x_1$ and $x_2$ is obtained as follows:

$$\frac{\partial^2 U}{\partial x_1 \partial x_2} = \frac{\partial^2 V}{\partial x_1 \partial x_2} = 0 \quad (3.B.5)$$

As expected, the value taken by attribute $x_2$ does not influence the marginal utility provided by attribute $x_1$. However, the results are quite different for the logit model. Cross-partial effects are obtained from the second derivate of the logit model with respect to $x_1$ and $x_2$:

$$\frac{\partial^2 F}{\partial x_1 \partial x_2} = \frac{\partial^2 F}{\partial V^2}\frac{\partial V}{\partial x_2}\frac{\partial V}{\partial x_1} + \frac{\partial F}{\partial V}\frac{\partial^2 V}{\partial x_1 \partial x_2} = \frac{e^V(e^V-1)}{(1+e^V)^3}(\beta_{21} + 2\beta_{22}x_2)(\beta_{11} + 2\beta_{12}x_1) + \frac{e^V}{(1+e^V)^2} \quad (3.B.6)$$

Although the cross-partial utility evidences no relationship between $x_1$ and $x_2$, the estimation process has indeed considered it. Even when interaction terms are not included in the model, there are some *confusing effects* which reveal this relationship between variables. Thus, even though no interaction terms are included in the linear utility function definition, its estimation considers the relationship between variables.

### *3.B.1.2. Model with interaction terms*

*Model definition*

A multiplicative interaction term between attributes $x_1$ and $x_2$ has been added to the model defined in Equation 3.B.1. Thus, given the deterministic utility $V$, which can be dependent on two independent and continuous attributes $x_1, x_2$, and their squared and interaction terms, the utility ($U$) can be written as follows:

$$U(x_1, x_2) = V(x_1, x_2) + \varepsilon = \beta_{11}x_1 + \beta_{12}x_1^2 + \beta_{21}x_2 + \beta_{22}x_2^2 + \beta_3 x_1 x_2 + \varepsilon \quad (3.B.7)$$

Similarly, $F$ is the logistic expression which is used to estimate indirectly the linear deterministic utility function $V$:

$$F = \frac{1}{1+e^{-V}} = \frac{1}{1+e^{-(\beta_{11}x_1+\beta_{12}x_1^2+\beta_{21}x_2+\beta_{22}x_2^2+\beta_3 x_1 x_2)}} \quad (3.B.8)$$

*Marginal effects*

Again, marginal effects are obtained, but now considering the inclusion of an interaction term between the attributes. Using Equation 3.B.7, the marginal utility is just the first derivative of the utility ($U$) with respect to each continuous explanatory variable $x_1$:

$$\frac{\partial U}{\partial x_1} = \frac{\partial V}{\partial x_1} = \beta_{11} + 2\beta_{12}x_1 + \beta_3 x_2 \quad (3.B.9)$$

The marginal utility of attribute $x_1$ depends also on the value taken by the attribute $x_2$. This interdependence between the utility provided by $x_1$ and $x_2$ will happen only if interaction terms are included in the utility function.

Hence, independently of the linear or logit utility function, the interdependence between $x_1$ and $x_2$ is considered as:

$$\frac{\partial F}{\partial x_1} = \frac{\partial F}{\partial V}\frac{\partial V}{\partial x_1} = \frac{e^V}{(1+e^V)^2}(\beta_{11} + 2\beta_{12}x_1 + \beta_3 x_2) \quad (3.B.10)$$

*Cross-partial effects*

Now, the inclusion of an interaction term between attributes recognises that the marginal utility of an attribute depends on the rest of the attributes. But, how does a change in the value of an attribute change the marginal utility of another attribute? The second derivate of the utility with respect to $x_1$ and $x_2$ illustrates this cross-partial utility:

$$\frac{\partial^2 U}{\partial x_1 \partial x_2} = \frac{\partial^2 V}{\partial x_1 \partial x_2} = \beta_3 \quad (3.B.11)$$

This reveals that a change in the value of $x_2$ causes a change in the marginal utility provided by $x_1$, which is quantified in $\beta_3$. Again, when cross-partial effects are calculated from the logit model, this relationship is also included:

$$\frac{\partial^2 F}{\partial x_1 \partial x_2} = \frac{\partial^2 F}{\partial V^2}\frac{\partial V}{\partial x_2}\frac{\partial V}{\partial x_1} + \frac{\partial F}{\partial V}\frac{\partial^2 V}{\partial x_1 \partial x_2} = \frac{e^V(e^V - 1)}{(1+e^V)^3}(\beta_{21} + 2\beta_{22}x_2)(\beta_{11} + 2\beta_{12}x_1) + \frac{e^V}{(1+e^V)^2}\beta_3 \quad (3.B.12)$$

Now, the cross-partial derivate depends on two separate types of effect: *confusing effects*, which are represented by the first part of the derivative, and *interaction effects*, which are represented by the last part of the derivative. In this way, the inclusion of an interaction term in the definition of the utility function changes the cross-partial effects, for both the linear utility function and the logit approximation.

### 3.B.2. Application

To better understand the proposed theoretical framework, the choice experiment described in the manuscript is used. The marginal and cross-partial utilities have been checked in the manuscript, focusing on the linear utility specification. Here, the assessment is centred on the relationship among utility, food provision (FOOD) ($x_1$) and water supply for irrigation (WATER) ($x_2$), using the logit model specification. The rest of the attributes included in the experimental design are considered as fixed. Two models have been employed. The first corresponds to Model 2 (*MXL-Quadratic*) in the paper - that is, the model without interaction terms - whilst the second refers to Model 3 (*MXL-All interactions*), the model with interaction terms.

Figure 3.B.1 shows the expected changes in the logistic expression (upper panel) and marginal effects (lower panel) of food provision when water supply changes, considering both Model 2 (left panel) and Model 3 (right panel). It points out that, even when no interaction terms are included in the utility function definition (Model 2), the logit model is able to capture the interrelationship between FOOD and WATER. In this way, although the two attributes are considered as independent in the choice experiment design, the estimation process actually takes into account the interrelationship between them. Therefore, some *confusing effects* do exist. Moreover, when the interaction term is included, both the marginal and cross-partial effects decrease, which evidences not only the presence of *interactions effects*, but also the decrease in the *confusing effects*.

The *confusing effects* reveal that the relationship between FOOD and WATER is negative, which is supported by the results shown in the manuscript. On the one hand, Figure 3.B.1 shows that the lower the food provision, the greater the marginal effects, independently of the level of the water supply for irrigation. This means that supplying water for irrigation has greater impacts on the marginal utility of FOOD when food provision is low. On the other hand, the cross-partial effects show that the impact of the water supply for irrigation on the marginal effects of food provision would be different depending on the amount of food provided. When FOOD is low, cross-partial effects would be positive if, and only if, the water supply for irrigation was high. This means that, when the level of food provision is low, supplying water for irrigation has a positive impact on the marginal utility of food provision, since it is implicitly understood that higher levels of water supply for irrigation are needed to increase food provision. However, if food provision is high, people get utility from this food provision level and, thus, higher levels of water supply for irrigation reduce the marginal effects of food provision. Therefore, when FOOD is high, cross-partial effects would be positive if, and only if, the water supply for irrigation was low.

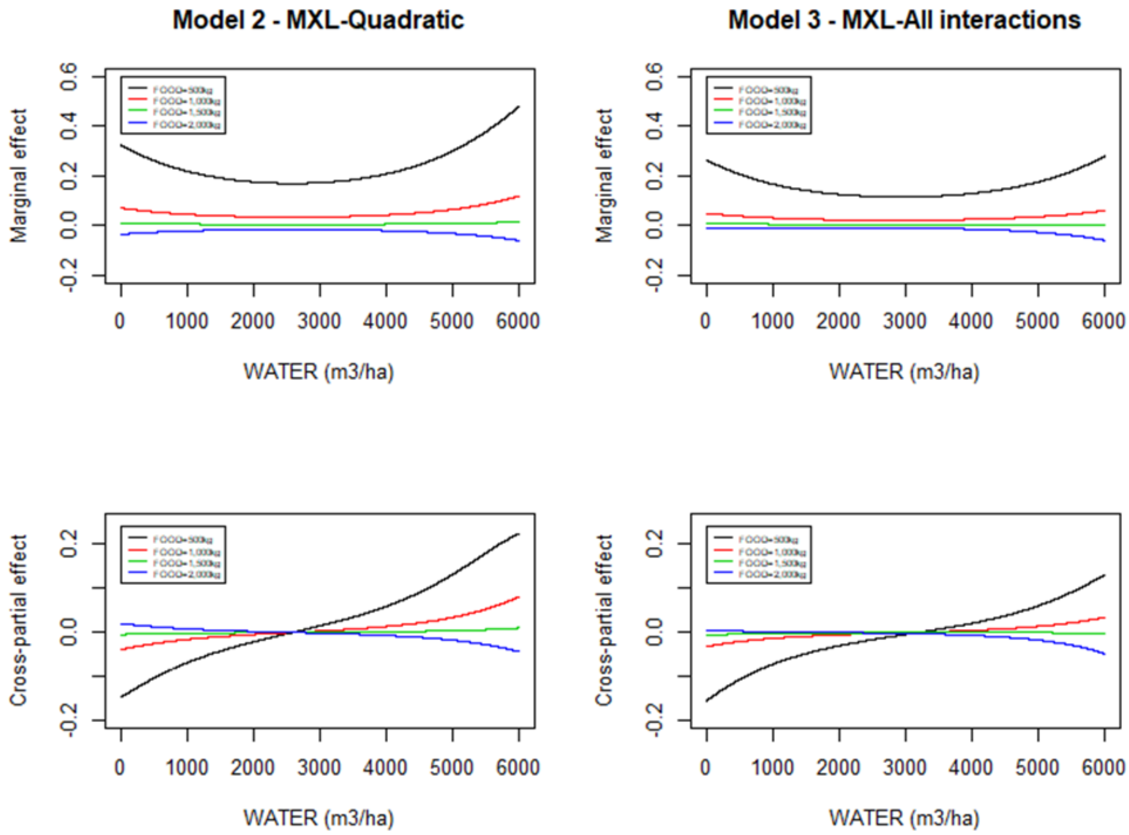


Figure 3.B.1. Marginal and cross-partial effects of FOOD.

Note: Marginal and cross-partial effects of FOOD obtained from logit model specifications. FOOD represents food provision and WATER is the water supply for irrigation. Both effects depend on the level of FOOD and WATER. The FOOD levels are represented by different-coloured lines. WATER is summarised in the X axis.

# Chapter 4. Assessment of social demand heterogeneity to inform agricultural diffuse pollution mitigation policies

This chapter is the accepted version of the article:

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## Highlights

- Social demand for agricultural nitrate pollution mitigation measures is estimated.
- Heterogeneous preferences for agricultural measures are disentangled.
- Socioeconomic benefits derived from the agricultural measures exceed their costs.
- Economic support is revealed when the good ecological status of surrounding ecosystems is achieved.

## Abstract

Agroecosystems provide several agroecosystem disservices, among which diffuse nutrient pollution is one of the most significant, mainly due to its negative impacts on surrounding ecosystems, such as coastal ecosystems. Therefore, the implementation of agricultural measures to mitigate nutrient pollution might become a way to overcome this environmental challenge. However, proper implementation requires both a cost-effectiveness assessment and social support. This paper aims to value the social demand for agricultural measures to mitigate nutrient pollution and the benefits gained in the surrounding ecosystems from their implementation. Additionally, the demand preference heterogeneity is assessed. Measures proposed by a law intended to mitigate diffuse nitrate pollution in the Campo de Cartagena catchment area (south-eastern Spain) and thereby restore one of the main coastal lagoons in the European Mediterranean Sea, the Mar Menor, are used as a case study. A choice experiment and latent class mixed logit were employed to disentangle heterogeneous social preferences. Despite the fact that preference heterogeneity was revealed regarding the proposed agricultural measures, strong preferences linked to the restoration of the Mar Menor were shown by all the distinct classes. The socioeconomic benefits derived from the measures along with the expected environmental benefits from the restoration of the surrounding ecosystems exceed the farmers' costs for their implementation. Consequently, the results provide guidance to policy makers in the establishment of socially supported strategies for agricultural nitrate pollution mitigation.

Keywords: Agriculture; Mitigation measures; Choice experiment; Preference heterogeneity; Latent class mixed logit.

## 4.1. Introduction

Agroecosystems produce positive outcomes, namely agroecosystem services (AES), which comprise food provision, climate regulation, biodiversity protection and even landscapes for enjoying leisure and recreation (Power, 2010). However, these ecosystems also provide negative outputs to society, or agroecosystem disservices (AEDS)[4], such as rivalry for the use of water resources and the generation of many sources of pollution, mainly in intensively-irrigated agroecosystems (Pajewski et al., 2020). One of these negative contributions is specifically nutrient pollution, which impacts many other surrounding ecosystems, such as wetlands and rivers (Monteagudo et al., 2012), groundwater (Lerner and Harris, 2009) and coastal landscapes (Lunau et al., 2013). Agricultural nitrate pollution induces several environmental issues, ranging from salinisation to eutrophication, which cause a decline in the health status of these surrounding ecosystems. The impact of nutrient pollution is such that it may be translated into a depletion of the ecosystem services provided by these ecosystems, thus reducing their ecological and socioeconomic value (Del Arco et al., 2015).

The degradation of aquatic ecosystems due to nutrient pollution is a critical issue worldwide and it may well increase in the next few years due to the increment in food demand, the consequent intensification of agriculture and the negative effects of climate change (WWAP, 2018). The challenge is, therefore, to manage efficiently the negative outcomes from agriculture due to the excessive nutrients in run-off and leachates from irrigated agroecosystems (Wu et al., 2020). The European Union established the Water Framework Directive (WFD) (Dir 2000/60/EC) and the Nitrate Directive (Dir 91/676/CEE) to achieve a good ecological status[5] of water bodies across Europe, as well as to determine and assess the management measures required for this. These have been translated finally into the establishment of nitrate vulnerable zones, as well as the creation of programmes of measures which comprise voluntary and mandatory practices for farmers to reduce the negative agricultural impact on water bodies.

Measures and strategies to mitigate nutrient pollution may be implemented to guarantee the sustainability of the agroecosystems and surrounding ecosystems (Geng and Sharpley, 2019).

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[4] Agroecosystem disservices (AEDS) are defined as the agroecosystems' "generated functions, processes and attributes that result in perceived or actual negative impacts on human wellbeing" (Shackleton et al., 2016). Consequently, they may comprise negative externalities, but are not restricted to them.

[5] According to the WFD, a good ecological status of water bodies is reached when "the values of the biological quality elements [...] show low levels of distortion resulting from human activity but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions".

Improvement of nutrient management practices, better irrigation strategies and adoption of new agricultural management practices, such as crop diversification, cover crops and green manure, are concrete measures to mitigate nutrient pollution (Christianson et al., 2014; Cui et al., 2020). Most of these measures may require deep changes in farm management and thus higher costs for farmers (Alcon et al., 2021). Hence, the selection and adoption of nutrient pollution mitigation measures need specific criteria to ensure their proportionality. On the supply side, cost-effectiveness has been widely applied to guide the selection and adaptation of these kinds of measures, based on comparing the expected effectiveness of the measures in terms of nutrient pollution abatement with the expected cost of their implementation (Balana et al., 2011). They also could be assessed in terms of farmers' acceptance when they involve changes in their agricultural management practices (Alcon et al., 2021). However, the benefits they provide are perceived by the entire society (Glenk et al., 2011), thereby revealing the importance of broadening the analysis to include the demand side.

Agricultural measures to mitigate nutrient pollution may improve not only the water quality of surrounding ecosystems but also the provision of AES. Agroecosystems, whose core activity is agriculture, impact human wellbeing by means of the provisioning, regulating and cultural services they provide. The provision of AES thereby depends on the agroecosystem functioning and capacity, where agricultural practices represent the main pressures for agroecosystems (Zabala et al., 2021a). Hence, agricultural measures to mitigate nutrient pollution are expected to influence agroecosystems management and thereby the provision of AES. For instance, among the above-mentioned measures, better irrigation practices may reduce the water use in irrigation, providing a positive effect on utility, while crop diversification or cover crops may increase carbon sequestration and biodiversity and enhance agricultural landscapes, also providing an increase in utility (Alcon et al., 2020). In this sense, AES are translated into economic benefits when they are valued by market and non-market valuation methods (Zabala et al., 2021b). Therefore, the implementation of agricultural nutrient mitigation measures may impact on the wellbeing through the resulting changes in AES, in addition to the expected improvement in water quality in surrounding ecosystems.

Environmental protection and restoration require policy mechanisms to ensure that measures are effectively implemented. Regulatory and incentive-based instruments represent the main categories of such policy mechanisms. While the former are based on controlling and limiting the actions that farmers might carry out, mainly through legislation, incentive-based instruments persuade farmers to implement such measures by using economic instruments that would increase the economic efficiency (Hahn, 2000). Fees, taxes, subsidies and tradeable permits are common examples of these economic instruments. The idea is that the adoption of these mechanisms by policy makers will be encouraged through the evaluation of the benefits and costs of the measures they are intended to help to implement. Hence, measures to mitigate agricultural nutrient pollution, which might imply not only private costs, but also benefits for the entire society, should be identified together with the costs, ensuring the principle of proportionality of the costs (Martin-Ortega, 2012).

The evaluation of costs and benefits considers, thereby, both the costs for farmers for implementing the measures and the benefits that society may obtain from both the reaching of a good ecological status and the implementation of the measures themselves. Public participation is, in this sense, three-fold (Glenk et al., 2011): (1) the costs of implementing the measures are met by farmers; (2) the benefits from reaching a good ecological status are perceived by the society, as are those obtained from the changes in AES derived from the measures themselves; (3) consequently, since they may involve public money investments, the preferences of both farmers and society as a whole for the different measures should be evaluated through cost-benefit analysis prior to their use in policy making, guaranteeing public expenditure acceptability.

The economic valuation has been a common key tool to assess nutrient pollution mitigation strategies. The classical cost-effectiveness analysis (CEA) uses the economic cost value, together with the physical effectiveness, as the basis for its implementation, but without considering the measured benefits. Hence, CEA focuses on non-monetary outcomes. In order to identify the benefits, the analysis of the demand should be developed with an economic approach. Most benefits related to nutrient pollution mitigation measures are due to improvements in the water quality of aquatic ecosystems, and are not directly traded in the markets. Thus, non-market valuation approaches, such as stated preference methods, are required to provide the economic value of such improvements (Rolfe and Windle, 2011). Besides, benefits can be estimated for specific measures on the basis of their social demand, given the utility people might obtain from their implementation. Then, cost-benefits analysis (CBA) will be more useful than CEA for evaluating the adoption of nutrient pollution mitigation strategies, especially when they affect water bodies for public use (Feuillette et al., 2016). Therefore, CBA focuses on monetizing the benefits and costs of such policy interventions.

In this context, this paper aims to value the social demand for agricultural measures to mitigate diffuse nutrient pollution and the benefits gained in surrounding ecosystems due to their implementation. Additionally, the demand preference heterogeneity is also assessed. For this purpose, the Campo de Cartagena catchment area and the Mar Menor coastal lagoon (SE Spain) were used as the case study, and a choice experiment as the core methodology. This assessment combines the social demand for the measures that are implemented in the agricultural sector in the Campo de Cartagena catchment area, to mitigate the diffuse nitrate pollution impact, with the expected socioeconomic benefits of reaching a good ecological status of the Mar Menor coastal lagoon. The use of a choice experiment ensures public involvement in public budget management, as well as shedding light on how much citizens are willing to pay to reach a good ecological status of an environmental asset and their preferences for the proposed measures (Glenk et al., 2011).

It is quite common to find, in the literature, non-market valuations for the benefits of water quality improvements, mainly regarding the application of the WFD. Hanley et al. (2006) valued the benefits of improving river water quality by diminishing agricultural non-point nitrate pollution in two catchment areas in eastern Scotland. Kataria et al. (2012) focused specifically on the Odensen river (Denmark) to value the benefits of its restoration. In addition, Hampson et al. (2017) centred their study on the water quality for recreation activities in the Yare river (England), while Andersen et al. (2019) focused on the nitrate concentration in the Danish coastal waters and the benefits associated with tourism and real estate value due to water quality improvements. Also, the benefits of water quantity improvements have been the object of valuation in water-scarce areas (Alcon et al., 2010, 2011; Berbel et al., 2011). All these works assessed the expected economic benefits from a demand perspective without analysing social preferences for the concrete measures needed to achieve water quality improvement. In contrast, specific measures intended to reach a good ecological status of water bodies have been mostly assessed from a supply perspective; that is, considering farmers' preferences and analysing their effectiveness, both biophysically and economically. Balana et al. (2011) remarked on the relevance of CEA in the assessment of water pollution mitigation strategies, while Balana et al. (2015) established that the catchment scale is appropriate for CEA and that agricultural measures are required to achieve a good ecological status of water bodies. Other initiatives broaden the scope - to address the factors and attitudes which determine whether farmers adopt water pollution mitigation measures (Inman et al., 2018), evaluate result-based payments schemes intended to reduce agricultural diffuse pollution, in SE Sweden (Sidemo-Holm et al., 2018), or assess the acceptability and perceived costs for farmers regarding the establishment of diffuse nitrate pollution mitigation measures (Alcon et al., 2021). However, although the global benefits associated with the improvement in water bodies quality have been assessed from a demand perspective, to the best of our knowledge, no work has analysed which specific measures are demanded by society. It seems that the social demand for the measures to reduce diffuse pollution of water has been disregarded in the literature, despite the fact that it could be a significant driver for their implementation and success and that these measures may imply social costs and benefits and public expenditure (Smith et al., 2017). In addition, they are also key drivers of changes in the provision of AES, increasing or decreasing human wellbeing, which adds to the significance of considering the social demand for their implementation. Although the environmental benefits of agricultural measures are centred on the mitigation of nitrate pollution, they also provide other environmental benefits, such as carbon sequestration, cooling effects and an increase in biodiversity (Arata et al., 2020). Hence, the novelty of this paper lies in the valuation of the social demand for specific measures (to be adopted by farmers) designed to mitigate agricultural nitrate pollution, as well as in the assessment of the preference heterogeneity in this demand.

Preference heterogeneity assessment is key to the public involvement in and the success of diffuse pollution mitigation measures. Understanding the factors that motivate the social demand for this type of measure allows policy makers to design agricultural policies which anticipate social support (Ren et al., 2020). Preference heterogeneity regarding benefits from water quality improvements may arise from sociodemographic and attitudinal individual characteristics, such as age, income (Chen and Ting Cho, 2019) or the contact with the environmental good reaching the good status (Kosenius, 2010; Hampson et al., 2017), and even from the spatial distribution of the benefits (Brouwer et al., 2010). Preference heterogeneity can be modelled in choice experiments by using mixed logit (MXL) models and latent class (LC) models. MXL models assume randomly distributed parameters, and LC models group individuals among classes according to their preferences. Also, a latent class mixed logit (LC-MXL) model blends both approaches to address heterogeneity.

This case study will help our understanding of preference heterogeneity, due to its socioecological characteristics. Nutrient loading into the Mar Menor was self-regulated, historically, until the summer of 2016, when the coastal lagoon passed its threshold point, precipitating a eutrophication crisis. An algal and phytoplankton bloom turned the water turbid and greenish, reducing drastically the water quality in the lagoon (Pérez-Ruzafa et al., 2019). This severe situation rapidly became apparent to the general public. Encouraged by NGOs and other stakeholders, coverage in the social media and press quickly expanded the concerns about the irreversible environmental damage suffered by the lagoon, with claims for responsibility to be accepted at the administrative and political levels (Perni et al., 2020). In consequence, the focus moved to the agricultural sector, blamed by society for being the main cause of the degradation of the Mar Menor. This situation, added to the fact that both agriculture and the Mar Menor are of significance for the regional population given their socioeconomic and ecological importance, increased the controversy of the problem. In addition, the degradation of the coastal lagoon has coexisted with the delay in the implementation of the WFD and continuous changes in the regional law on nitrate pollution mitigation in the last few years (Perni et al., 2020). The existing law, intended to produce a good ecological status of the Mar Menor coastal lagoon, focuses on regulating the agricultural activity in its catchment area, and has become a controversial topic for society, with supporters and detractors and thereby heterogeneous opinions regarding which measures should be implemented to restore the ecosystem. Hence, the paper contributes to the assessment of the preferences of the public in their support for agriculture-related measures to mitigate diffuse nitrate pollution, in which preference heterogeneity is a key issue to address. The results will also contribute to the design of the measures demanded to mitigate agricultural nitrate pollution as well as to the expected benefits of their implementation. This will involve the society in the public budget management, which, combined with the expected cost of the measures, means that the specific cost-benefits will be important when considering the measures to adopt. Agri-environmental projects are expected to be better supported by local communities if the beneficiaries are deeply involved (Sarvilinna et al., 2018).

The paper is structured as follows. In the following section the case study and methods are presented, including the choice experiment design, its implementation and the econometric framework followed. Section 4.3 presents the main results, whilst Section 4.4 discusses them and their implications for policy design. Finally, Section 4.5 concludes the paper.

## 4.2. Methodology

### 4.2.1. Case study

The case study is located in the Campo de Cartagena catchment area in south-eastern Spain (Figure 4.1), within the Segura River Basin District. It has a Mediterranean semiarid climate, with an average annual rainfall of 300 mm, and includes 169,450 ha of agricultural land. The main irrigated area is the 'Campo de Cartagena' Irrigation Community, which integrates intensive, modern and precise agriculture, yielding fruit and vegetables with high added value.

The Campo de Cartagena catchment area finally discharges in the Mar Menor, the largest hypersaline coastal lagoon in Europe. It covers 135 km$^2$ and is separated from the Mediterranean Sea by a sand bar that is 20 km long and between 100 and 900 m wide. The Mar Menor contains unique habitats, and so is protected at the international level: Natura 2000, Ramsar Wetland, Specially Protected Area of Mediterranean Importance, among others (Perni et al., 2011).

Its environmental importance makes the Mar Menor a singular ecosystem to be preserved and protected from its main pressures, which include agriculture, tourism, mining and fishing (Velasco et al., 2018). The lagoon receives the runoffs from the Campo de Cartagena basin in several ephemeral watercourses (ramblas) which transport nutrient-enriched water and sediments from the surrounding crop-growing areas and mining sites. To illustrate this, among the surface watercourses, the Rambla del Albujón is the most important. It has a flow of around 7.3 hm$^3$/year, with a load of 219 t/year of nitrate and 52 t/year of total phosphorus (García-Pintado et al., 2007). More than 50% of this nitrate discharge comes from agricultural sources, the value being below 30% for phosphates. Besides, the groundwater in the catchment area, which also drains to the Mar Menor, is highly saline due to the presence of excess nutrients from agriculture. Groundwater discharges are estimated to be between 40 $hm^3$/year (Domingo-Pinillos et al., 2018) and 78 $hm^3$/year (Alcolea et al., 2019), which means that between 1,400 t/year and 10,200 t/year of nitrate reach the lagoon. All these discharges, together with the insufficient wastewater treatment capacity and the massive tourist influx, have resulted in an increase in the nutrient concentration in the lagoon, finally leading to eutrophication and the generation of algal blooms. This situation peaked in 2016, when the eutrophication and algal blooms processes worsened, changing the colour of the water, increasing its turbidity and reducing considerably the benthic habitats (Pérez-Ruzafa et al., 2019).

This situation, together with the delay in the WFD implementation, focused public opinion on the Mar Menor management, resulting in protests demanding real conservation and restoration actions and political responsibilities (Perni et al., 2020). This strong socio-political debate concluded with the approval of Law 1/2018, on 7 February 2018, regarding urgent measures to ensure the environmental sustainability of the Mar Menor and the surrounding areas (BORM, 2018). This law, still under debate, and slightly modified by Law 3/2020, on 27 July 2020, regarding the integral recovery and protection of the Mar Menor (BORM, 2020), establishes a set of the main mandatory agricultural measures to be implemented by farmers with the purpose of, among other goals, reducing nutrient run-off and leaching from crop fields in the Campo de Cartagena catchment area to the Mar Menor. The main measures are as follows[6]:

- A ban on vegetable crops less than 100 m from the coastline ("farmland >100m coastline").
- Installation of a system to reduce the nitrate content of the water obtained from desalination plants prior to its use in crops and its entry into groundwater and/or the Mar Menor ("denitrification plants").
- Establishment of hedgerows of native plants around farm perimeters ("perimeter hedgerow").
- Compliance with a good agricultural practices (GAP) code based on the efficient use of fertilisers and irrigation water, in accordance with the Nitrates Directive ("GAP code").

[6] See Alcon et al. (2021) for an in-depth description of the measures.

Against expectations, the approval of this Law did not appease public concerns about the ecological status of the Mar Menor, and even generated an increase in the divergence of the positions, between those who consider these measures necessary but not enough to preserve and restore the coastal lagoon (Guaita-García et al., 2020) and those who questioned their intensity and mandatory nature (e.g. the agricultural sector) (Martínez-Álvarez and Martin-Gorriz, 2018).

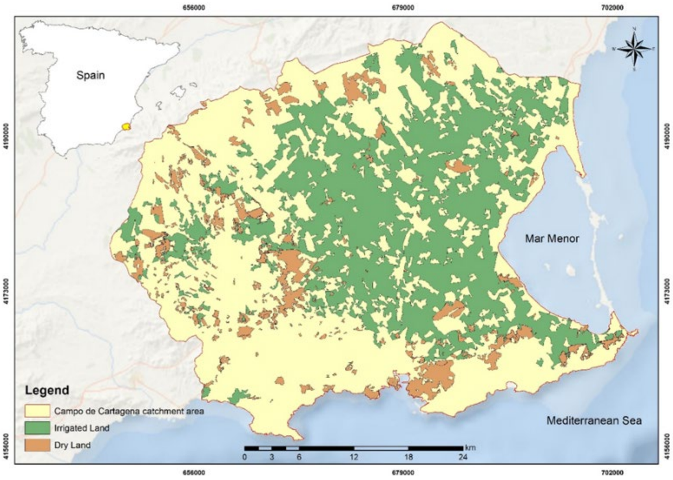


Figure 4.1. Case study.

### 4.2.2. Choice experiment method

The choice experiment is a stated preference method based on the multi-attribute utility (Lancaster, 1966) and random utility (McFadden, 1974) theories. In a discrete choice experiment an individual is asked to choose among different alternatives. An alternative, defined by a set of attributes, is chosen when it provides the highest utility level. The choices are then assessed to disentangle the individuals' preferences, based on a utilitarianism approach (Champ et al., 2017). The inclusion of a monetary attribute allows estimation of the economic value of the attributes. The choice experiment was selected as an appropriate method to estimate the social demand for agricultural measures and water quality improvement since it allows one to measure values one-by-one and the trade-offs among them.

#### *4.2.2.1. Experimental design*

Law 1/2018 represents the starting point for this work. Hence, the relevant attributes to be valued were selected and defined on the basis of this Law, as mandatory measures for the agricultural sector to deal with nitrate pollution. Table 4.1 shows the attributes and levels included in the choice experiment, which comprise the aforementioned agricultural measures in addition to the expected water quality improvement. Water quality improvement was expressed as a categorical attribute with three levels: no improvement, moderate improvement and high improvement. These levels were defined according to the biological quality indicators established by the WFD. Therefore, two qualitative indicators - regarding (1) the presence of algae and phytoplankton and (2) changes in the diversity and abundance of benthic habitats - were used. The current situation (no improvement) is defined by assuming a great presence of algae and phytoplankton and a decline in benthic habitats; a moderate improvement in water quality is reached with a medium presence of algae and phytoplankton, while no changes are expected in the diversity and abundance of benthic habitats; and a great improvement in water quality is defined by a low presence of algae and phytoplankton and an increase in the diversity and abundance of benthic habitats in the medium term.

In this case study, water quality is a multi-factorial issue. The coexistence of different pressures on the Mar Menor means that we cannot isolate the effects of the agricultural measures intended to improve its ecological status. However, the implementation of agricultural measures to mitigate diffuse nitrate pollution, ceteris paribus, does improve the ecological status of the case study area (Alcolea et al., 2019). The improvement in water quality due to the application of some of the established measures could range from moderate to high since a linear relationship between the measures and the water quality improvement could not be defined. Besides, the intensity in the implementation of the agricultural measures also determines their effectiveness in improving water quality. As Martínez-Paz et al. (2013) showed, agricultural measures, together with other, non-agricultural actions, such as the improvement in urban wastewater treatments, are expected to improve water quality in the Mar Menor in such a way that it reaches a good ecological status. Hence, each measure cannot be associated with a specific improvement in the water quality, but every improvement in the water quality can be reached with every measure. This ensures that the implementation of at least one agricultural measure is needed to improve the water quality of the Mar Menor. But, above all, it guarantees that the attribute levels of the water quality improvement and the agricultural measures are not correlated in terms of the design.

Table 4.1. Attributes and levels in the choice experiment.

| Attribute | Definition | Levels |
| --- | --- | --- |
| Water quality improvement | Improvement in water quality (presence of algae and phytoplankton \| benthic habits) in the Mar Menor due to the application of some of the agricultural measures established by Law 1/2018 | No improvement (SQ)<br>Moderate improvement (baseline)<br>High improvement |
| Agricultural measures | Agricultural measures established by Law 1/2018 to ensure the environmental sustainability of the Mar Menor, and consequently to improve its water quality | No measure (SQ)<br>Farmland >100m coastline (baseline)<br>Denitrification plants<br>Perimeter hedgerow<br>GAP code |
| Cost (€/household/year) | Taxes reallocated to support the agricultural measures for the next five years | 10<br>20<br>30<br>40 |

Agricultural measures to mitigate nitrate pollution provide other environmental benefits beyond the reduction of agricultural diffuse pollution. These benefits encompass an enhancement of the provision of AES (Zabala et al., 2021a), and therefore are expected to impact also on human wellbeing (Zabala et al., 2021b). Apart from the mitigation of the agroecosystem disservice of nitrate pollution, the agricultural measures considered within the experimental design are expected to have positive and negative impacts on the provision of provisioning, regulating and cultural AES, which adds trade-offs to the assessment of these measures. For instance, the establishment of a perimeter hedgerow increases biodiversity (Heath et al., 2017) and improves landscape values (Dachary-Bernard and Rambonilaza, 2012). Table 4.2 shows the expected impacts on AES, identified according to a literature review.

Table 4.2. Impact of agricultural measures to mitigate nitrate pollution on agroecosystem services (AES).

| Agricultural measures | AES [a] | AES category [a] | References |
|---|---|---|---|
| Farmland >100m coastline | (-) Food | (-) Provisioning AES<br>(=) Regulating AES<br>(=) Cultural AES | Zabala et al. (2021b) |
| Denitrification plants | (+) Water | (+) Provisioning AES<br>(=) Regulating AES<br>(=) Cultural AES | Alcon et al. (2021) |
| Perimeter hedgerow | (+) Biodiversity<br>(+) Aesthetic values (landscape) | (=) Provisioning AES<br>(+) Regulating AES<br>(+) Cultural AES | Heath et al. (2017)<br>Assandri et al. (2016)<br>Dachary-Bernard and Rambonilaza (2012) |
| GAP code | (+) Biodiversity<br>(-) Erosion | (=) Provisioning AES<br>(+) Regulating AES<br>(=) Cultural AES | Almagro et al. (2016) |

[a] Expected change in AES are summarised as reduction (-), increase (+) or maintenance (=).

The cost attribute was defined as the yearly household taxes reallocated (Rogers et al., 2020) over the next five years to support and monitor the implementation of these agricultural measures by farmers.

The attribute levels were combined using an S-efficient design, which minimizes the sample size required to estimate significant parameter values, and employing the Ngene 1.0.2 software package (Rose et al., 2010). The priors were obtained from a 15-respondent pre-test choice experiment. The final design comprised 16 choice sets grouped in 4 blocks. Each block was randomly distributed to each respondent during the survey, thereby each respondent faced 4 choice sets. Each choice set was composed of 3 alternatives, one representing the current situation, or status quo (SQ), and two others. An example of a choice set is shown in Figure 4.2. The SQ alternative was the situation where no agricultural measure was implemented, and therefore water quality would not improve. Given the case study features and so the experimental design, the SQ attribute levels could not be included in the rest of the alternatives because the achievement of water quality improvement without any measure is not rational, nor is the application of a measure without any quality improvement. Hence, baseline levels were employed to allow the preference modelling. In the case of the water quality improvement attribute, the moderate level was used as the baseline, while the prohibition of farmland less than 100 m from the coastline was employed regarding the agricultural measure attribute. The latter was selected as the baseline since it is the measure which has the lowest impact in the overall case study area, affecting an insignificant area within the catchment area.

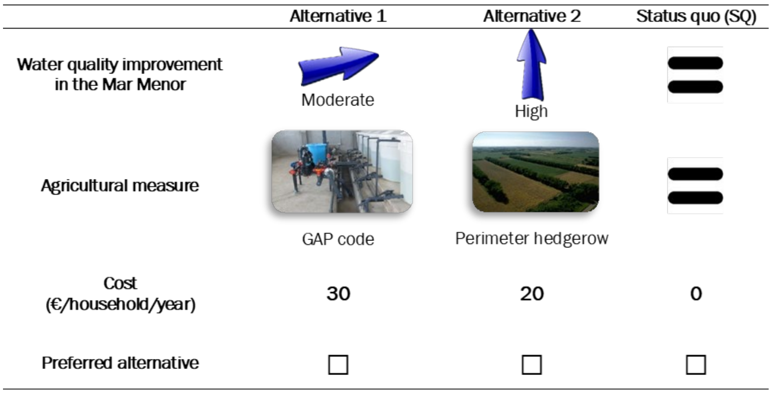


Figure 4.2. Example of a choice set.

### *4.2.2.2. Econometric framework*

According to the random utility theory (McFadden, 1974), the utility ($U_{ijt}$) provided for an individual *i* by the choice of an alternative *j* in a choice set *t* can be decomposed into an observed ($V_{ijt}$) and an unobserved part ($\varepsilon_{ijt}$), considered additively:

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt} = \beta_{iSQ} SQ_t + \sum_{k=1}^{K} \beta_{ik} X_{kjt} + \varepsilon_{ijt} \tag{4.1}$$

Where $V_{ijt}$ is the deterministic part of the utility, determined by the SQ dummy alternative and the *k* attribute levels ($X_{kjt}$), and $\varepsilon_{ijt}$ is a stochastic error term, identically and independently distributed following a Gumbell-distribution. Assuming $V_{ijt}$ to be a weighted sum of the attribute levels, $\beta_{iSQ}$ and $\beta_{ik}$ represent, respectively, the individual marginal utility obtained from the SQ alternative and each *k = 1, ..., K* attribute, reflecting the expected changes in utility derived from water quality improvement and the agricultural measures.

The multinomial logit (MNL) model is the one applied most widely to assess choice preferences. However, it assumes fixed coefficients for all the individuals, namely, homogeneous preferences across individuals. To overcome this issue, preference heterogeneity has traditionally been analysed by two main separate approaches. First, preference heterogeneity is treated as a taste variation among individuals by allowing the marginal utility to change among them. The mixed logit (MXL) model, which assumes that the coefficients follow a continuous distribution across individuals, is the most feasible way to disentangle preference heterogeneity in such cases (Train, 2009). Alternatively, individuals can be categorised into a set of groups or classes with similar within-class taste preferences. The latent class (LC) models, which assume that the population consists of groups of individuals that are homogeneous within each class but differ among classes, are employed to understand this preference heterogeneity approach (Greene and Hensher, 2003). Hence, the challenge is to link both types of heterogeneity, within-class and across classes. The LC-MXL model is employed for this (Greene and Hensher, 2013).

The LC-MXL model assumes that individuals are distributed heterogeneously in the population, with a discrete distribution, into a finite number $S$ of latent classes, while taste variation across individuals within each class is allowed (by coefficients) to follow a continuous distribution. The probabilities to be allocated into classes are determined by using the individuals' characteristics $z_i$ (e.g. socio-demographic characteristics, environmental attitudes, relationship with the good valued ...). Hence, the probability of the individual $i$ residing in class $s$ can be written as:

$$Prob\ (class = s) = \pi_{is}(\theta, z_i) = \frac{e^{\theta'_s z_i}}{\sum_{s=1}^{S} e^{\theta'_s z_i}},\ s = 1, \dots, S;\ \theta_S = 0 \tag{4.2}$$

The traditional LC model, which assumes fixed coefficients within each class, is therefore broadened to include within-class heterogeneity. This type of preference heterogeneity is measured as (Greene and Hensher, 2013):

$$\beta_{ik|s} = \beta_{ik} + w_{ik|s} \tag{4.3}$$

$$w_{ik|s} \sim N(0, \sigma_{ik|s}) \tag{4.4}$$

Hence, in the current study, the LC-MXL model was estimated as follows:

$$U_{ijt|s} = V_{ijt|s} + \varepsilon_{ijt|s} = \beta_{iSQ|s} SQ_t + \sum_{k=1}^{K} \beta_{ik|s} X_{kjt} + \varepsilon_{ijt|s}\ ,\quad s = 1, \dots, S \tag{4.5}$$

Where $\beta_{iSQ|s}$ and $\beta_{ik|s}$ are the marginal utility provided by the SQ alternative and attribute  , respectively, for individual $i$ in class $s$. This not only permits the modelling of unobserved heterogeneity across individuals, but also allows the independence of irrelevant alternatives (IIA) to be overcome (Hensher et al., 2005). The LC-MXL model was estimated using the maximum simulated likelihood estimator (Train, 2009). Specifically, we modelled the utility function in R software (R Core Team, 2019), using the Apollo package (Hess and Palma, 2019) and 500 Sobol draws with Owen scrambling for the simulation of the log-likelihood function (Owen, 1995). The number of latent classes was selected by evaluating the goodness-of-fit of each model on the basis of the likelihood value at convergence, the Akaike Information Criterion (AIC) and the Bayesian Information Criterion (BIC).

The economic value of the water quality improvement and the agricultural measures was estimated using the marginal rate of substitution (MRS) between the non-cost attributes and the cost attribute, which shows the willingness to pay (WTP). Following on from Equation 4.5, the WTP was calculated as follows:

$$MRS_c^{ik}|s = WTP_{ik|s} = \frac{\partial U_{ijts}/\partial X_{kjt}}{\partial U_{ijts}/\partial COST} = -\left(\frac{\beta_{ik|s}}{\beta_{ic|s}}\right) \tag{4.6}$$

Where $\beta_{ic|s}$ refers to the marginal utility of the cost attribute for class $s$, and $WTP_{ik|s}$ represents, in monetary terms, how much each individual $i$ in class $s$ is willing to pay to improve the water quality in the Mar Menor or to support each agricultural measure $k$. However, the WTP does not represent the main tool for policy making due to the presence of the SQ. Compensating surplus (CS) which measures the change in wellbeing when moving from the current situation to another one with better water quality and the implementation of an agricultural measure, is thus a more appropriate instrument to guide policy decisions. It also quantifies how much individuals are willing to pay to support concrete measures and to restore the ecological status of the coastal lagoon. The CS can be derived using the following Hanemann utility difference formula (Hanemann, 1984):

$$CS_{im|s} = -\frac{1}{\beta_{ic|s}}\left[ln\left(\sum e^{V_{im|s}}\right) - ln\left(\sum e^{V_{isq|s}}\right)\right] \tag{4.7}$$

Where $CS_{im|s}$ represents the consumer surplus derived from changing from the current situation (SQ) to a specific management scenario $m$ where a certain level of water quality improvement has been reached and a specific agricultural measure has been implemented. Hence, $V_{isq|s}$ is the utility obtained in the current situation (SQ) for individual $i$ in class $s$, while $V_{im|s}$ is the utility derived from a water quality improvement and the implementation of an agricultural measure. Positive values for $CS_{im|s}$ represent people who get positive utility changes from the change to a restored situation in the Mar Menor, and hence they are willing to pay to support this improvement.

### 4.2.3. Sampling and data collection

The choice experiment was developed through a survey, using a questionnaire with three main sections. The first comprised questions addressing the relationship between the respondents and the Mar Menor, including a five-point Likert scale to look into the respondents' opinions and attitudes about the pressures responsible for the degradation of this coastal lagoon. The second comprised the choice experiment. In this section, a brief description of the purpose of the survey was included, as well as the main points about the implementation of the agricultural measures and their expected impacts on water quality improvement in the Mar

Menor. The respondents were also shown an example of a choice set: it was explained to them what it represented and what they needed to do. Then, in order to mitigate hypothetical bias, a cheap talk was employed together with a budget reminder and a reminder about the opportunity cost that their choices may imply in terms of social welfare (Penn and Hu, 2019; Borgar et al., 2021). If public money is reallocated to the implementation of the agricultural measures, and no increase in the public budget is expected, this means a reduction in public funds for other sectors, such as education and health. In addition, the respondents were informed that their responses will be used by policy makers to design publicly supported measures (Zawojska et al., 2019). For those who always chose the SQ alternative, a follow-up question was included to address the reasons for this behaviour and to disentangle protest responses. The last section of the questionnaire included questions on the environmental attitudes and sociodemographic characteristics (sex, age, monthly income…) of the respondents. The environmental attitudes were measured by an ecological commitment index, using a five-point Likert scale to evaluate a set of statements in the questionnaire[7].

The data were collected, through a face-to-face survey, by trained numerators in summer 2019. The target population comprised not only the Campo de Cartagena basin, but also the rest of the Region of Murcia (543,800 households), since the expected benefits of water quality improvements are perceived not only in the coastal neighbourhoods but also by the rest of the citizens of the region, given the socioecological importance of the lagoon. The survey was implemented in public places - such as parks, markets, shopping centres and even waiting rooms for the renewal of ID cards and for family doctors - where, *a priori*, the probability of being there is the same for all people over 18 years of age. Given that the payment vehicle was tax reallocation, only those respondents who were in charge –responsible or co-responsible- of paying household taxes were surveyed. Hence, 576 households comprised the final sample.

Table 4.3 shows the main descriptive statistics of the sample. The sample was representative of the regional census data in terms of gender, monthly income and educational level, which ensures the results represent social preferences. Of the respondents, 74% stated that they had visited the Mar Menor at least once in the last year, making them users of the lagoon. Furthermore, the respondents were asked to rate the main activities which may be responsible for the degradation of the Mar Menor. They considered wastewater discharges, agriculture and urban development to be the main pressures, which shows the social concerns about the need to implement agricultural measures to prevent further degradation of this ecosystem and to restore it. This also reveals that the respondents were aware of the importance of

[7] See Alcon et al. (2019) and Perni et al. (2020) for further information about this index.

implementing agricultural measures to mitigate nitrate pollution, in order to tackle the ecological status of the Mar Menor, and thereby the on-going impact of these measures on the improvement of the water quality in the case study area.

Table 4.3. Sample descriptive statistics.

| Variable | | Sample | Region of Murcia | |
|---|---|---|---|---|
| *Sociodemographic information* | | | | t-test (p-value) |
| Age (years) | | 38.56 | 48.25[a] | -14.03 (0.00) |
| Gender (% women) | | 48.61 | 50.39[a] | 0.02 (0.89) |
| Household income (€/month) | | 2,087 | 2,037[b] | 0.87 (0.39) |
| Educational level (%) | | | | Pearson $\chi^2$ (p-value) |
| | Lower education | 7.64 | 3.30[c] | 8.32 (0.04) |
| | Primary education | 18.75 | 16.30[c] | |
| | Secondary education | 26.56 | 46.70[c] | |
| | Higher education | 47.05 | 33.70[c] | |
| Environmental commitment (EC) | | 4.22 | | |
| *Relation to the Mar Menor* | | | | |
| User (people visited the Mar Menor in the last year) (%) | | 73.96 | | |
| How do you consider the following activities impact on the ecological status of the Mar Menor? | | | | |
| (1 No impact – 5 High impact) | Wastewater discharges | 4.38 | | |
| | Agriculture | 4.13 | | |
| | Urban development | 3.91 | | |
| | Coastal infrastructures | 3.55 | | |
| | Mining | 3.45 | | |
| | Tourism | 3.22 | | |
| | Fishing | 2.86 | | |
| | Boating and sport | 2.83 | | |

[a] INE (2019a); [b] INE (2019b); [c] INE (2019c)

## 4.3. Results

### 4.3.1. Estimated choice models

The presence of protest respondents was assessed before applying the econometric models. Protest respondents comprise those who refuse to participate in the hypothetical market or do not approve the survey design or implementation. Hence, they are not willing to pay or, if they are, they provide underestimated economic valuations (Barreiro-Hurle et al., 2018). Here, the protest respondents were identified as those individuals who chose the SQ alternative in all the choice-sets and also stated that "Public Administration must be in charge of the improvement and maintenance costs" or "It should be funded by the polluters". Out of the 576 respondents, 88 (15.28%) revealed protest behaviour, leaving a final sample of 488 respondents for the estimation of the choice models. A binary logit model was applied to disentangle the sociodemographic and attitudinal factors that led to protest responses. This showed that protest behaviour might be explained by educational levels alone, since the probability of giving a protest response was lower for people with higher educational levels.

The utility function was modelled using different specifications. Table 4.4 reports the best-fitted models. From Model 1 to Model 5, the modelling of social demand heterogeneity becomes increasingly complex. Thus, Model 1 shows the simplest specification, namely an MNL model, whilst Model 2 possesses an MXL specification with random, coefficient-based preference heterogeneity. The last three models represent LC models with two latent classes: Model 3 includes non-random heterogeneity, Model 4 adds randomly distributed coefficients and Model 5 includes covariates to explain the class allocation. The number of classes was determined using the AIC.

Model 1 shows that most of the agricultural measures are socially demanded. Although the Hausman test (Hausman and McFadden, 1984) did not allow rejection of the IIA hypothesis ($\chi 2 = 9.42$, p value = 0.15), the MNL specification was considered inappropriate to model the preferences for water quality and agricultural measures since it cannot capture unobserved preference heterogeneity. Therefore, an MXL model was estimated in Model 2. All the attributes and the SQ alternative were modelled randomly considering a Normal distribution, while the cost attribute followed a log-normal distribution. It is of note that all the coefficients are significant and have the expected signs. The significance of the SD coefficients reveals the presence of preference heterogeneity among the respondents. Besides, the negative sign of the coefficient of the SQ alternative reflects the disutility that people get from the current situation of the Mar Menor, and thus the desire to improve its current ecological status. This is in accordance with the positive sign of the mean coefficients of the agricultural measures, which shows that, on average, the application of all these agricultural measures is socially acceptable. As expected, the Log-Likelihood Ratio (LR) test (LR = 396.19; p-value = 0.00), the AIC and the BIC confirmed that Model 2 performed better than Model 1.

However, as discussed above, preference heterogeneity could be assessed by assuming different groups or classes of respondents sharing similar preferences. Model 3 is an LC model with 2 classes, but includes within-class homogeneous preferences. In terms of performance, Model 3 fitted better than Model 1 (LR = 259.86; p-value = 0.00), but not Model 2, according to the assessment using the AIC and BIC. Thus, the presence of latent classes and the existence of taste variation among individuals could not be disregarded, and so were assessed integrally in the present choice models. Hence, Models 4 and 5 describe LC-MXL models, which merge both types of preference heterogeneity, showing 2 classes and within-class random parameters for all attributes and the SQ alternative. Indeed, Model 4 performed better than Model 2 (LR = 60.34; p-value = 0.00) and Model 3 (LR = 196.67; p-value = 0.00), showing the need to consider both types of preference heterogeneity. Model 5 adds sociodemographic and attitudinal variables to Model 4 to explain the class allocation model, specifically accounting for *user*, *EC*, *age*, *household income*, *higher education* and the stated expected impact of agricultural activity on the ecological status of the Mar Menor (*Impact-Agri*) (see Table 4.3). Model 5 performed better than Model 4 (LR = 10.74; p-value = 0.06), indicating that differences in preference between classes can be explained by some sociodemographic and attitudinal variables; therefore, it will be used for further assessment.

Table 4.4. Econometric model results.

| Model | Model 1 | Model 2 | Model 3 | | Model 4 | | Model 5 | |
|---|---|---|---|---|---|---|---|---|
| Specification | MNL | MXL | LC | | LC-MXL (1) | | LC-MXL (2) | |
| | | | Class 1 | Class 2 | Class 1 | Class 2 | Class 1 | Class 2 |
| | Coef. (S.E.) | Coef. (S.E.) | Coef. (S.E.) | Coef. (S.E.) | Coef. (S.E.) | Coef. (S.E.) | Coef. (S.E.) | Coef. (S.E.) |
| **Utility function** | | | | | | | | |
| *Mean* | | | | | | | | |
| SQ | -1.79 *** | -3.76 *** | -3.52 *** | 0.02 | -3.90 *** | -7.04 *** | -4.08 *** | -8.37 *** |
| | (0.16) | (0.35) | (0.41) | (0.31) | (0.71) | (1.17) | (0.24) | (1.73) |
| Water quality | 0.48 *** | 0.71 *** | 0.59 *** | -0.14 | 0.28 * | 2.56 *** | 0.24 * | 4.83 *** |
| High improvement | (0.07) | (0.13) | (0.09) | (0.28) | (0.16) | (0.77) | (0.14) | (0.59) |
| Agricultural measure | | | | | | | | |
| Denitrification plants | 0.21 ** | 0.29 * | 0.20 * | 0.46 | 1.25 *** | -2.87 ** | 1.02 *** | -5.25 *** |
| | (0.10) | (0.17) | (0.12) | (0.32) | (0.29) | (1.25) | (0.21) | (0.93) |
| Perimeter hedgerow | 0.20 ** | 0.49 *** | 0.18 * | 0.12 | 1.02 *** | -1.07 | 0.84 *** | -1.60 |
| | (0.08) | (0.13) | (0.10) | (0.47) | (0.24) | (0.68) | (0.18) | (1.22) |
| GAP code | 0.10 | 0.15 | 0.19 ** | -0.55 * | 1.02 *** | -2.22 ** | 0.76 *** | -3.56 *** |
| | (0.08) | (0.12) | (0.10) | (0.33) | (0.21) | (1.08) | (0.15) | (1.38) |
| Cost | -0.03 *** | -0.07[a] *** | -0.04 *** | | -0.09[a] *** | | -0.09 *** | |
| | (0.01) | (0.01) | (0.01) | | (0.02) | | (0.01) | |
| *SD* | | | | | | | | |
| SQ | | 0.50 | | | -1.74 *** | 3.10 *** | -1.91 *** | 3.34 *** |
| | | (0.50) | | | (0.64) | (0.79) | (0.50) | (1.28) |
| Water quality | | -1.38 *** | | | -0.73 ** | -1.50 ** | -0.73 ** | 1.91 * |
| High improvement | | (0.20) | | | (0.36) | (0.76) | (0.31) | (1.03) |
| Agricultural measure | | | | | | | | |
| Denitrification plants | | 1.66 *** | | | -0.81 | 3.91 *** | 0.91 | 6.52 *** |
| | | (0.42) | | | (0.67) | (1.26) | (0.58) | (1.77) |
| Perimeter hedgerow | | 0.75 * | | | 0.13 | 3.48 *** | 0.01 | -6.26 *** |
| | | (0.45) | | | (0.63) | (1.07) | (0.73) | (1.37) |
| GAP code | | 1.19 *** | | | 0.32 | 2.99 *** | -0.25 | 5.85 *** |
| | | (0.19) | | | (0.55) | (0.98) | (0.66) | (0.66) |
| Cost | | 0.15[a] *** | | | 0.19[a] *** | | 0.18 *** | |
| | | (0.04) | | | (0.07) | | (0.04) | |
| **Class allocation** | | | | | | | | |
| Probability | | | 0.84 | 0.16 | 0.60 | 0.40 | 0.67 | 0.33 |
| Constant | | | -1.66 *** | | 0.40 | | 5.30 *** | |
| | | | (0.24) | | (0.34) | | (1.48) | |
| User | | | | | | | 0.04 | |
| | | | | | | | (0.37) | |
| EC | | | | | | | -0.56 ** | |
| | | | | | | | (0.24) | |
| Age | | | | | | | -0.01 | |
| | | | | | | | (0.01) | |
| Higher education | | | | | | | -0.41 | |
| | | | | | | | (0.39) | |
| Impact-Agri | | | | | | | -0.36 ** | |
| | | | | | | | (0.16) | |
| **Model description** | | | | | | | | |
| Log-Likelihood | -1,838.64 | -1,640.55 | -1,708.70 | | -1,610.38 | | -1,605.01 | |
| Adjusted-R2 | 0.14 | 0.23 | 0.20 | | 0.24 | | 0.24 | |
| AIC | 3,689.29 | 3,305.10 | 3,441.43 | | 3,264.76 | | 3,264.02 | |
| BIC | 3,722.75 | 3,372.02 | 3,508.35 | | 3,422.16 | | 3,414.59 | |

Statistically significant at a level of *0.1, **0.05 or ***0.01.

[a] Cost coefficients are assumed to follow a log-normal distribution. Mean ($\beta_{c|s}$) and standard deviations ($\sigma_{c|s}$) cost coefficients reported are corrected by using $\beta_{c|s} = e^{(b_c+s_c^2/2)}$ and $\sigma_{c|s} = e^{(b_c+s_c^2/2)} \times \sqrt{e^{s_c^S} - 1}$, where $b_c$ and $s_c$ are the mean and standard deviation of the natural logarithm of the cost coefficient.

Model 5 represents a two-class LC-MXL model with class allocation regressors. The mean coefficients are significant at the 10% level or higher, showing their relevance to the explanation of the utility function. Only the mean coefficient of the perimeter hedgerow level for class 2 is found not significant. The negative sign of the mean coefficient for the SQ alternative in both classes discloses the disutility people obtain from the current situation of the Mar Menor. Much needs to be done to restore the ecological status of the coastal lagoon and this is demanded by all the citizens. These statements are in line with the results for the achievement of a high level of water quality, whose coefficients show a positive sign in both classes. However, divergences between classes arise concerning the preferred agricultural measures. Whilst class 1 receives positive utility from the implementation of each measure, compared to banning farmland less than 100 m from the coastline (the baseline), individuals from class 2 get disutility from their implementation, revealing that they prefer that the measures not be applied, in comparison to banning farmland along the coastline. Moreover, preference heterogeneity between classes seems to arise not only in the sign of the coefficients, but also in the size of the expected utility impacts. The expected impact on the disutility due to the SQ alternative is higher for class 2, as is the utility that individuals in class 2 get from higher water quality.

The significance of the SD coefficients depends on the class: the SD coefficients for water quality and SQ are significant in both classes, while the SD coefficients for the agricultural measures are significant in class 2 but not in class 1. This again reveals preference heterogeneity between the classes: individuals in class 1 have homogeneous preferences regarding the utility they get from implementation of the agricultural measures, while the respondents in class 2 display great divergence in their preferences concerning the agricultural measures.

In view of the above, Model 5 shows two latent classes well defined by their taste preferences. Class 1 comprises 67% of the respondents - namely, those who get some disutility from the current situation, desire the water quality of the Mar Menor to be improved to some extent and, above all, support all the proposed agricultural measures to be implemented, compared with the banning of farmland near the coast. In contrast, class 2, representing 33% of the respondents, shows stronger preferences for all the attributes, higher disutility from the SQ alternative, higher utility at high water quality and greater disutility from the implementation of each agricultural measure, compared with the utility from banning farmland. These remarkable differences in expected utility could be related to the attitudinal characteristics of the respondents, specifically their EC and the stated expected impact of agricultural activity on the ecological status of the Mar Menor. Therefore, the negative sign of their coefficients in the class allocation models shows that the greater the EC or the greater the expected impact of agriculture, the greater the probability of an individual belonging to class 2. In fact, each increment in EC increases the probability of belonging to class 2 by 9.82 percentage points, on average, and by 7.24 points in the case of Impact-agri. Only attitudinal variables are relevant to explain the class allocation, whilst individual sociodemographic characteristics, such as age, income or education level, cannot be considered to disentangle the preferences for diffuse nitrate pollution mitigation policies. Hence, people with greater ecological commitment are more concerned about the current situation of the Mar Menor and also desire greatly that this situation improves. Moreover, the people who think that agriculture is mainly responsible for the degradation of the lagoon also consider that these agricultural measures are not good enough as banning farmland less than 100 m from the coastline, to restore the ecosystem.

*Post-hoc* analysis of the sociodemographic and attitudinal variables of both classes corroborated the results of the class probability model within Model 5. Significant differences between classes were found only in terms of the educational level, EC and perceived impact of different activities on the ecological status of the Mar Menor. Therefore, respondents belonging to Class 2 exhibit higher educational levels and greater pro-environmental behaviour as well as perceiving a greater impact of such pressures on the ecological status of the Mar Menor, mainly for the agricultural pressure, which shows the greatest difference between the classes. Table 4.5 shows these descriptive statistics, as well as the tests applied to verify the differences between classes.

Table 4.5. Descriptive statistics by classes.

| Variable | | Class 1 | Class 2 | |
|---|---|---|---|---|
| *Sociodemographic information* | | | | t-test (p-value) |
| Age (years) | | 37.97 | 38.27 | -0.18 (0.86) |
| Gender (% women) | | 45.32 | 52.74 | -1.50 (0.13) |
| Household income (€/month) | | 2,076 | 2,061 | 0.11 (0.91) |
| Educational level (%) | | | | Pearson $\chi^2$ (p-value) |
| | Lower education | 6.14 | 7.53 | 12.92 (0.01) |
| | Primary education | 21.05 | 18.49 | |
| | Secondary education | 28.07 | 21.23 | |
| | Higher education | 44.74 | 60.27 | t-test (p-value) |
| Environmental commitment (EC) | | 4.11 | 4.50 | -5.16 (0.00) |
| *Relation to the Mar Menor* | | | | |
| User (people who visited the Mar Menor in the last year) (%) | | 74.56 | 74.66 | -0.02 (0.98) |
| How do you consider the following activities impact on the ecological status of the Mar Menor? | | | | |
| (1 No impact – 5 High impact) | Wastewater discharges | 4.30 | 4.48 | -2.17 (0.03) |
| | Agriculture | 3.96 | 4.53 | -5.29 (0.00) |
| | Urban development | 3.77 | 4.13 | -3.40 (0.00) |
| | Coastal infrastructures | 3.49 | 3.68 | -1.75 (0.08) |
| | Mining | 3.46 | 3.55 | -0.67 (0.50) |
| | Tourism | 3.20 | 3.36 | -1.42 (0.16) |
| | Fishing | 2.80 | 3.11 | -2.78 (0.01) |
| | Boating and sport | 2.83 | 2.86 | -0.23 (0.82) |

### 4.3.2. Economic valuation of water quality improvement and agricultural measures

The preferences for the water quality and agricultural measures determine the WTP. Table 4.6 shows the WTP values. Positive WTP values reveal the monetary WTP that people are willing to reallocate to get a high water quality and to implement the proposed measures, whilst negative values exhibit the economic compensation expected for the maintenance of the current situation, or the implementation of some measure, as in the case of Model 5 - class 2. Besides, the average values, which consider the class allocation shares, are also presented.

Table 4.6. Marginal WTP for water quality improvement and agricultural measures (€/household/year) [95% confidence interval].

| | Class 1 | Class 2 | Average [a] |
|---|---|---|---|
| SQ | -80.98 | -154.88 | -105.37 |
| | [-84.41; -77.55] | [-161.79; -147.98] | [-109.91; -100.83] |
| Water quality | | | |
| High improvement | 6.48 | 79.70 | 30.64 |
| | [5.82; 7.13] | [76.17; 83.23] | [29.19; 32.09] |
| Agricultural measure | | | |
| Denitrification plants | 18.19 | -30.85 | 2.01 |
| | [17.12; 19.26] | [-41.16; -20.55] | [-1.93; 5.94] |
| Perimeter hedgerow | 15.74 | -11.08 | 6.89 |
| | [15.13; 16.35] | [-20.02; -2.13] | [3.92; 9.86] |
| GAP code | 14.07 | -11.34 | 5.69 |
| | [13.50; 14.65] | [-18.83; -3.84] | [3.00; 8.37] |

[a] Weighted WTP for the attribute levels estimated by considering the class probabilities.

As expected from the results for Model 5, both classes show a negative WTP for SQ and positive values for high water quality improvement. However, the values are greater for class 2. Whilst class 1 is willing to pay 80.98 €/household/year for leaving the SQ, class 2 is willing to pay 154.88 €/household/year. Moreover, to get high-quality water, class 1 is willing to pay only 6.48 €/household/year, while class 2 is willing to pay 79.70 €/household/year. These differences will also determine the CS values of the proposed agricultural management scenario, which are greater for class 2 respondents due to their stronger preferences for water quality improvement and abandonment of the SQ.

Regarding the specific agricultural measures to mitigate nitrate pollution, class 1 is willing to pay for every measure, whose values range from 14.07 to 18.19 €/household/year, the use of denitrification plants being the most supported measure. By contrast, class 2 shows negative WTP values, revealing the disutility its members obtain from the implementation of such measures compared to the banning of farmland less than 100 m from the coastline. The adoption of denitrification plants was the worst-valued measure for class 2, followed by the GAP code and perimeter hedgerow, respectively. Notwithstanding, averaged across both classes, all the agricultural measures had a positive WTP, which shows that their implementation may enhance social wellbeing.

However, when the purpose is to inform policy design, CS becomes a more proper and realistic indicator, since it not only summarises the value of specific measures, but also includes the WTP associated with changes from the current situation.

The CS shows the wellbeing gain due to the water quality improvement, relative to the current situation of degradation, achieved by the implementation of each agricultural measure. Different agricultural management scenarios are proposed to account for the expected impact of the implementation of the measures and the improvement in water quality on wellbeing. Table 4.7 shows these proposed scenarios, which are defined according to their expected effect on the provision of AES.

Table 4.7. Definition of agricultural management scenarios to mitigate nitrate pollution.

| | Water quality improvement [a] | Agricultural measures | Agroecosystem services (AES) [b] |
|---|---|---|---|
| Scenario 1<br>Base scenario | (1a) Moderate<br>(1b) High | Farmland >100m<br>coastline | (-) Provisioning AES<br>(=) Regulating AES<br>(=) Cultural AES |
| Scenario 2<br>Provisioning-based | (2a) Moderate<br>(2b) High | Denitrification plants | (=) Provisioning AES<br>(=) Regulating AES<br>(=) Cultural AES |
| Scenario 3<br>Regulating-based | (3a) Moderate<br>(3b) High | Perimeter hedgerow<br>GAP code | (=) Provisioning AES<br>(+) Regulating AES<br>(+) Cultural AES |
| Scenario 4<br>In-between proposal | (4a) Moderate<br>(4b) High | Denitrification plants<br>Perimeter hedgerow | (=) Provisioning AES<br>(+) Regulating AES<br>(+) Cultural AES |

[a] Both levels of water quality improvement can be reached within each scenario.
[b] Expected changes in AES are summarised as reduction (-), increase (+) or maintenance (=).

The implementation of the proposed scenarios would improve the water quality to some degree, at least to the "moderate" level. Therefore, as stated in the experimental design section, both levels of water quality improvement can be reached, and so the CS values of both these situations were estimated per scenario. Table 4.8 summarises the CS values by scenario, by class and by the average across them. Independently of the water quality improvement reached, greater CS values were obtained from scenario 4 for the respondents of class 1, since their most valued agricultural measures are implemented, and from scenario 1 for class 2, since these respondents prefer that no agricultural measures are applied. Therefore, scenario 3 becomes, *ceteris paribus*, the combination of agricultural measures that provides the greatest social wellbeing to society: nearly 118 €/household/year and 150 €/household/year for moderate and high water quality improvements, respectively.

Table 4.8. CS for agricultural management scenarios to mitigate nitrate pollution (€/household/year) [95% confidence interval].

| | Water quality improvement [a] | Class 1 | Class 2 | Average [b] |
|---|---|---|---|---|
| Scenario 1 | (1a) Moderate | 80.98<br>[77.55; 84.41] | 154.88<br>[147.98; 161.79] | 105.37<br>[100.83; 109.91] |
| | (1b) High | 87.46<br>[83.80; 91.12] | 234.58<br>[224.58; 244.59] | 136.01<br>[130.29; 141.73] |
| Scenario 2 | (2a) Moderate | 99.17<br>[94.86; 103.48] | 124.03<br>[110.54; 137.53] | 107.37<br>[100.48; 114.27] |
| | (2b) High | 105.65<br>[101.16; 110.14] | 203.73<br>[188.22; 219.24] | 138.02<br>[130.23; 145.80] |
| Scenario 3 | (3a) Moderate | 110.80<br>[106.22; 115.38] | 132.47<br>[117.45; 147.49] | 117.95<br>[110.68; 125.22] |
| | (3b) High | 117.28<br>[112.49; 122.07] | 212.17<br>[195.19; 229.14] | 148.59<br>[140.39; 156.80] |
| Scenario 4 | (4a) Moderate | 114.91<br>[110.01; 119.81] | 112.96<br>[96.90; 129.02] | 114.27<br>[106.52; 112.01] |
| | (4b) High | 121.39<br>[116.31; 126.47] | 192.65<br>[174.97; 210.34] | 144.91<br>[136.33; 153.49] |

[a] Both levels of water quality improvement can be reached within each scenario.
[b] Weighted CS for the agricultural management scenarios estimated by considering the class probabilities.

## 4.4. Discussion

When AEDS affect surrounding ecosystems, their significance extends beyond the environmental impact, provoking social and economic issues (Zabala et al., 2021b). Thus, the implementation of management measures which mitigate the AEDS could be a way of overcome this challenge, and not only should the effectiveness of their implementation be assessed but also their demand and expected economic impacts. This is the case of agricultural diffuse pollution, specifically nitrate pollution, which has been studied here. The social demand for and economic value of measures to mitigate agricultural nitrate pollution have been estimated, together with the expected improvement in the water quality of the surrounding ecosystem: The Mar Menor coastal lagoon.

The results highlight the presence of heterogeneous social preferences. The degradation of the Mar Menor ecosystem has concerned public opinion increasingly in the last few years. This has been translated into the social demand for increased protection of the lagoon and the restoration of its ecological status. Moreover, the public debate has also turned to the measures that should be implemented to restore the ecosystem, which has put the agricultural sector in the spotlight, and there are both those in favour and those against the imposition of severe measures in the agricultural sector to reduce nitrate pollution. Hence, here, two latent classes have been obtained, well defined by their preferences regarding the agricultural measures to be implemented. The first class, which encompasses most of the population, shows positive preferences in the support of agricultural measures, whilst the second class, despite its stronger preference for high water quality improvement, does not endorse the implementation of such measures. The greater the environmental concern, the less the support for these agricultural measures but the greater the desire to improve the Mar Menor. This finding is supported by the work of Perni et al. (2011). Hence, the preferences of those who are more concerned about the environment, and think that agriculture is the main problem facing this lagoon, seem to reveal that they prefer a ban on farmland less than 100 m from the coast (baseline), instead of the rest of the measures, to mitigate agricultural nitrate pollution. This shows that the citizens see the degradation of the Mar Menor as a polarized conflict, where policy implementation has failed over the last few years (Perni et al., 2020).

The results show that most citizens support the proposed agricultural measures, whilst around a third of those surveyed stated that they were not in favour of such measures. The installation of denitrification plants was the most controversial measure, followed by compliance with the GAP code. These results are supported by Guaita-García et al. (2020), who showed that, although agricultural diffuse pollution was publicly considered the main pressure on the Mar Menor, the management measures demanded to improve its ecological status did not always involve a reduced input of agricultural nutrients. In fact, according to their results, 73% of the respondents supported measures to mitigate agricultural nitrate pollution, among which the installation of denitrification plants and the establishment of hedgerows were highlighted. In contrast, those who rejected these measures considered that management should focus on intervention measures within the lagoon.

The social demand for the conservation and restoration of the Mar Menor ecosystem is not a new topic in the literature. For instance, Perni et al. (2011) showed the public preferences with regard to reaching a good ecological status of the ecosystem by applying the WFD. Velasco et al. (2018) valued the ecosystem services provided by this coastal lagoon and revealed the WTP for its conservation. Hence, the restoration of the Mar Menor ecosystem is crucial for increasing the wellbeing of the citizens. The value we estimate here for the abandonment of the current situation - to reach at least a moderately good ecological status - ranges from 58.60 to 100.13 €/household/year, and rises to a maximum of 158.57 €/household/year for the achievement of high water quality. These values are in line with those obtained by Perni et al. (2011), who valued the moderate restoration of the lagoon at 60.62 €/household/year and a good ecological status at 106.52 €/household/year, and by Velasco et al. (2018), who showed that the WTP for the conservation of the Mar Menor was 89.55 €/household/year.

Nonetheless, this is the first time that specific measures have been economically valued from the demand side and, despite the heterogeneity shown, the social demand for their implementation is reflected. In addition to mitigating a significant agroecosystem disservice for the case study area, namely nutrient pollution (Zabala et al., 2021a), the proposed measures also are able to impact the provision of AES both positively and negatively, generating trade-offs in their analysis. The assessment of the social demand for the measures allows better understanding of how these trade-offs balance out, and whether other impacts on human wellbeing are also considered by respondents facing the challenge of diffuse nitrate pollution from agriculture. Again, preference heterogeneity highlights the differences between the two classes. Whilst class 1 considers positively the benefits from regulating and cultural AES, the measure that reduces provisioning AES being the least preferred, class 2 disregards the regulating and cultural AES that some of the measures may provide. Hence, the class 1 preferences lead to a win-win situation, at least in terms of the AES promoted.

However, the assessment of the social demand for the agricultural measures is not enough, and their supply needs to be assessed also. Alcon et al. (2021) analysed the farmers' willingness to implement the same measures proposed by Law 1/2018. They showed the perceived cost for farmers to be 98.47 €/ha/year for the installation of the denitrification plants, 103.93 €/ha/year for the establishment of a perimeter hedgerow and 134.93 €/ha/year for compliance with the GAP code. The farmers' preferences for the agricultural measures are in line with those revealed here for class 1 respondents, with a similar ranking of preferred nitrate mitigation measures: (1) denitrification plants, (2) perimeter hedgerow and (3) GAP code. Hence, both the farmers and class 1 respondents demand changes at the farm level, but prefer not to change the agricultural management practices, especially those related to the compliance with the GAP code and which comprise restrictions in the use of fertilisers and modification of current irrigation strategies. The farmers' acceptance of the agricultural measures is then boosted by the demands of the local community, helping policy advisors in their implementation (Vrain and Lovett, 2016). Therefore, a large part of the society supports the implementation of agricultural measures to mitigate agricultural nitrate pollution. However, the range of feasible policies and measures that could be applied when the issue concerns the degradation of a coastal ecosystem is so wide that it is difficult to reduce it to just agricultural ones. The Class 2 preferences, which are very strong with regard to reaching moderate and high water quality improvement, show that alternative measures could also be applied to mitigate diffuse nitrate pollution and improve the ecological status of surrounding ecosystems. At the farm level, variation of the main crops, better irrigation strategies (Chen et al., 2010; Kay et al., 2019) or new agricultural management practices, such as crop diversification, cover crops and green manure (Christianson et al., 2014; Cui et al., 2020), could be implemented to mitigate nitrate pollution. At the catchment scale, these measures could be broadened to include the improvement of the wastewater treatment plants, the expansion of the denitrification plants, the restoration of the main watercourses that discharge into the ecosystem (Perni and Martínez-Paz, 2013) or even the reduction of the irrigated area (Guaita-García et al., 2020). Also, concerning the specific case study of the Mar Menor, its restoration may imply the implementation of management measures for urban-tourism development, such as the improvement of the public transport network and infrastructures around the lagoon and restrictions (or a ban) on the construction of second residences and hotels (Guaita-García et al., 2020). Hence, multifactorial issues require multidimensional solutions that cannot be reduced to one factor.

The main challenges that the agri-environment sector will face in the coming decades comprise the sustainability of the agroecosystems and the efficient management of water resources, which include water quality and security (WWAP, 2018). In this context, these results can guide policy makers in the improvement of the design and establishment of socially supported agricultural nitrate mitigation policies. The assessment of preference heterogeneity provides some guidance to enhance the social acceptability of such agricultural measures, which is especially significant for the case study area. Farmers, in consonance with policy makers, need to tackle the damaged reputation that the agricultural sector has acquired in the last few years, given the increase in social concerns about environmental issues that has led to the expression of unfounded opinions that weaken the public confidence in the agricultural sector (Arcas-Lario et al., 2021). Environmentally-friendly agricultural practices that integrate agricultural development, the mitigation of AEDS and the enhancement of AES are key in this sense. Social support for the restoration of degraded ecosystems could also be increased by promoting pro-environmental behaviour by individuals. Raising public awareness of the importance of pro-environmental behaviour increases EC, thereby reinforcing social support for the improvement of environmental quality.

The benefits obtained from applying the proposed agricultural measures could be compared with the expected costs; therefore, the application of CBA becomes straightforward. Table 4.9 summarises the CBA applied to the proposed agricultural management scenarios, considering the economic value of their implementation and the expected improvement in the water quality of the Mar Menor. Results from Alcon et al. (2021) - who assessed the same agricultural measures from the supply side- have been built upon for this purpose. The annual equivalent cost (AEC) was employed to account for the costs of the agricultural measures. It includes the investment and maintenance costs of each measure, estimated by considering a period of five years and a discount of 3.5% (Alcon et al., 2021; Almansa and Martínez-Paz, 2011). The same specifications apply to the estimation of the annual equivalent benefits (AEB). To quantify the AEB, the CS values were transformed into spatial economic values aggregating the $CS_{im|s}$ over the target population (543,800 households) and distributing it across the irrigated land in the Campo de Cartagena catchment area (44,000 ha). The CBA was applied only to the irrigated farms in the case study, since most of these agricultural measures are hardly significant for rainfed farms. This is the case of the denitrification plants, which apply only to the brackish drainage water flowing from irrigated farms, and the GAP code, whose mandatory practices are based on the efficient use of fertilisers and irrigation water. Indeed, the perimeter hedgerow, which was initially mandatory for both irrigated and rainfed farms, was suppressed for rainfed farms in the last law reform (BORM, 2020).

The results from the CBA show that the benefits of all the agricultural management scenarios far exceed the farm costs of their implementation, providing social support for their implementation. Notwithstanding, significant differences exist among them. Scenario 3, which provides the greatest CS, and so the greatest AEB, is also associated with the highest AEC and the lowest B/C ratio, making it the least cost-efficient scenario. This contrasts with scenario 1, which provides the lowest CS and AEB, but, given its lower AEC, is the most cost-efficient scenario, whose B/C ratio rises to 137.45 when the water quality improvement is high. Scenarios 2 and 4 occupy the second and third positions, respectively, according to their cost-efficiency.

Table 4.9. Costs and benefits for agricultural management scenarios to mitigate nitrate pollution (€/ha/year).

| | Agricultural measures | AEC [a] | Water quality improvement [b] | AEB [c] Class 1 | Class 2 | Average [d] | B/C ratio [e] |
|---|---|---|---|---|---|---|---|
| Scenario 1 | Farmland >100m coastline | 12.23 | (1a) Moderate | 1,000.85 | 1,914.23 | 1,302.27 | 106.48 |
| | | | (1b) High | 1,080.90 | 2,899.22 | 1,680.96 | 137.45 |
| Scenario 2 | Denitrification plants | 70.00 | (2a) Moderate | 1,225.66 | 1,532.92 | 1,327.05 | 18.96 |
| | | | (2b) High | 1,305.71 | 2,517.90 | 1,705.75 | 24.37 |
| Scenario 3 | Perimeter hedgerow GAP code | 325.81 | (3a) Moderate | 1,369.37 | 1,637.22 | 1,457.77 | 4.47 |
| | | | (3b) High | 1,449.42 | 2,622.21 | 1,836.46 | 5.64 |
| Scenario 4 | Denitrification plants Perimeter hedgerow | 168.14 | (4a) Moderate | 1,420.23 | 1,396.03 | 1,412.24 | 8.40 |
| | | | (4b) High | 1,500.28 | 2,381.02 | 1,790.93 | 10.65 |

Note: AEC and AEB estimated for a 5-year period ($T$) and using a social discount rate ($i$) of 3.50%, considering 543,800 households and 44,000 ha of irrigated farms.

[a] AEC: Annual Equivalent Cost (for farmers). AEC spatially averaged for the Campo de Cartagena area considering crop types in its estimation. Source: Alcon et al. (2020).

[b] Both levels of water quality improvement can be reached within each scenario.

[c] AEB: Annual Equivalent Benefit. $AEB = NPB \frac{i}{1-(1+i)^{-T}}$, where $NPB = \sum_{t=1}^{T} \frac{CS_{m|s_t}}{(1+i)^t} \times \frac{H}{A}$ with NPB: Net Present Benefits, H: households and A: extension of irrigated farms.

[d] Weighted AEB for the agricultural management scenarios estimated by considering the class probabilities.

[e] $B/_C\ ratio = \frac{AEB}{AEC}$

The present results, along with those of Alcon et al. (2021), support the successful acceptance and implementation of the measures proposed to mitigate agricultural nitrate pollution, considering both their demand and supply. Therefore, for an efficient policy implementation, the following recommendations should be considered: (1) the proposed agricultural measures should be implemented, given the benefits they provide in terms of the water quality improvement and AES.; (2) the socioeconomic benefits derived from the social support for the measures, as revealed here, could be applied to finance a programme of aid to farmers, to ensure the implementation of the measures; hence, incentive-based mechanisms should be encouraged. For instance, subsides for implementing the agricultural measures could be developed. Besides, payment for ecosystem services (PES) to compensate the provision of AES could also be applied to incentivize the planting of hedgerows and the implementation of the

GAP code; (3) it should be ensured that the measures are effective with regard to reaching high water quality in the surrounding ecosystems, given the greater benefits derived from improved water quality.

## 4.5. Conclusions

The implementation of agricultural measures to mitigate agricultural nitrate diffuse pollution needs to expand the current cost-effectiveness framework to include the assessment of their social demand and value, particularly when surrounding coastal ecosystems are affected. The integration of socioeconomic values together with the environmental benefits allows the incorporation of all the expected impacts of the pollution mitigation measures, by disentangling social preferences to ensure the compliance of the public.

The presence of latent classes among the citizens has evinced the heterogeneous preferences regarding the implementation of these agricultural measures, but there is a strong desire that the surrounding ecosystem be restored and the water quality recovered. Thus, the agricultural measures to mitigate agricultural nitrate pollution are widely supported by their demand and the expected water quality improvement, which far exceed the costs of their implementation. Consequently, the implementation of these measures is justified by the socioeconomic benefits that they provide, and which generate financial resources to encourage farmers to adopt them and to compensate them for this.

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# Chapter 5. General discussion and conclusions

## 5.1. Overview and answer to research questions

The present thesis shows the adaptation of the main ecosystem service paradigms to the particular case of agroecosystem valuation. A comprehensive approach integrating positive and negative agroecosystem outcomes, namely, AES and AEDS, is proposed, and human wellbeing is therefore placed at the core of the valuation. This approach has been verified by agroecosystem stakeholders and applied for the actual non-market valuation of AES, AEDS and agricultural practices. Social demand for the most significant AES and AEDS identified by stakeholders was estimated by using a discrete choice experiment for the general population. The non-market value of AES and AEDS allowed us to identify the wellbeing impact of changes in the provision of AES and AEDS in monetary terms. Then, this frame was expanded to also include the social demand for agricultural practices intending to mitigate AEDS, in particular, nutrient pollution to surrounding water ecosystems. The focus of the thesis is on the Region of Murcia (south-eastern Spain), as a representative case study for the semiarid Mediterranean region.

The analysis of stakeholders' preferences for AES and AEDS has been used to validate a comprehensive approach for the valuation of the AES and AEDS provided by semiarid Mediterranean agroecosystems. This approach is based on the framework for anthropised ecosystems developed by Barot et al. (2017), and it adapted the main accepted ecosystem services paradigms: MEA (2005), TEEB (2010), and CICES (Haines-Young and Potschin, 2018) to the particular case of agroecosystems. The stakeholder assessment enabled us to determine which AES and AEDS should be relevant for an agroecosystem valuation. By using choice experiments, stakeholders dealt with, a priori, 9 AES (food, climate regulation, soil maintenance, biodiversity, resilience, cultural heritage, aesthetic landscape values, opportunities for recreation and tourism, cognitive development) and 3 AEDS (water, emissions of contaminants into the atmosphere, water pollution), among which ultimately 4 AES and 2 AEDS were selected as the most important. The stakeholders' choices revealed biodiversity (38%) as the most important of the AES to be valued, followed by recreation (20%), temperature regulation (7%), and food provision (5%). Among the AEDS, water supply for irrigation and groundwater pollution were considered of equal weight (at 15% each). The approach included at least one of the AES or AEDS from every category (provisioning, regulating, and cultural), in line with the multifunctional character of agricultural activity (Huang et al., 2015).

The results from the stakeholder assessment revealed that the valuation of agroecosystems needs to deal with both positive and negative outcomes. That is what was done. In particular, those AES and AEDS that were found significant for agricultural stakeholders were socially valued. Food, climate regulation, maintenance of genetic diversity and opportunity for recreation and tourism as AES, and water and water purification and waste treatment as AEDS, were primarily considered to be socially valued using a discrete choice experiment to the general population. The integrated valuation of AES and AEDS allowed us to estimate the social demand for their provision whilst their biophysical trade-offs were also translated to the socioeconomic viewpoint. The socioeconomic system recognised therefore the importance that the general population attaches to the wellbeing contribution of AES and AEDS. Thus, all the considered AES and AEDS were socially valued, positively or negatively depending on their social consideration about their impact on human wellbeing. The modelling results of the choice experiment revealed the existence of diminishing marginal and cross effects for some of the AES and AEDS, therefore showing that, contrary to expected, not all AES and AEDS are socially perceived as such. Furthermore, it also revealed that the actual trade-offs taking place in biophysical agroecosystems are perceived by the socioeconomic system. As such, food, water, climate regulation, maintenance of genetic diversity and opportunities for recreation and tourism revealed positive marginal utility, among which food and water showed diminishing marginal utility. Then, increasing food levels and irrigation water supply to agroecosystems are not always socially desired. In contrast, water pollution was effectively considered as one of the AEDS by the general population, also revealing diminishing marginal utility. It shows that the disutility that nutrient groundwater pollution provides increases at a higher rate than pollution does. In addition, cross effects between food and water, as well as between water and water pollution, reveal trade-offs in their social demand. The non-market value of food provision depends not only on the amount of food provided by agroecosystems but also on the amount of irrigation water supplied to them. Similarly, the socially perceived value of water supply for irrigation depends negatively on the amount of food provided by agroecosystems and on the level of agricultural nutrient pollution impacting surrounding water ecosystems.

Such disentangled relationships among the social demand of AES and AEDS make marginal WTP values dependent on their own provision levels and of other AES and AEDS. This is the reflection of the non-linear utility functions of AES and AEDS. Thus, marginal values may not be the best way to compare the relative importance of AES and AEDS for agroecosystem valuation. It will depend on the provision level they reach in each specific agroecosystem. For instance, the TEV of the rainfed, traditional irrigated and highly-intensive irrigated agroecosystem in the case study area rounds the figure of 816 €/ha/year, 1,208 €/ha/year and 612 €/ha/year, respectively. Among these TEV, they agree on food being the AES that provides the greatest contribution, followed by the maintenance of genetic biodiversity. However, it is important to note that the relatively reduced TEV of the highly-intensive irrigated agroecosystem is due to the negative contribution of water pollution. This evinces the need to reduce groundwater nutrient pollution due to agricultural outcomes to increase its non-market value. Agricultural practices to mitigate nutrient pollution are a direct consequence of the obtained results.

As it has been revealed, when AEDS affect surrounding ecosystems, their significance extends beyond the environmental impact, provoking social and economic issues. Thus, the implementation of management measures that mitigate AEDS could be a way of overcoming this challenge, and not only should the effectiveness of their implementation be assessed but also their demand and expected economic impacts. This is the case in agricultural diffuse pollution, specifically nitrate pollution, which has been studied here. The non-market value of measures to mitigate agricultural nitrate pollution have been estimated, together with the expected improvement in the water quality of a surrounding ecosystem, the Mar Menor coastal lagoon.

The results showed that most citizens supported the proposed agricultural measures. However, preferences were not homogeneous across society. As such, two latent classes were obtained, well defined by their preferences regarding the agricultural measures to be implemented. The first class, which encompasses two-thirds of the population, shows positive preferences in the support of agricultural measures, whilst the second class, despite its stronger preference for high water quality improvement, does not endorse the implementation of such measures. The installation of denitrification plants was the most controversial measure, given its non-market value for each class. The results, therefore, highlighted the presence of heterogeneous social preferences. The sources of preference heterogeneity which allowed us to identify latent classes comprehended environmental commitment and the perceived impact of agricultural pollution as being responsible for the degradation of the Mar Menor. The greater the environmental concern, the less the support for these agricultural measures but the greater the desire to improve the Mar Menor. Nevertheless, despite preference heterogeneity, benefits from implementing agricultural measures to tackle nitrate pollution far exceeded the costs for farmers, suggesting policy makers the adequacy of their support. It also revealed the need for additional (agricultural and non-agricultural) measures to restore water quality in the Mar Menor to comply with those not supporting the proposed ones.

In light of the results obtained, it can be affirmed that the central aim of the thesis, which was *to value economically the integrated social demand of AES and AEDS and the agricultural practices that promote them by adopting a comprehensive approach for agroecosystem valuation in the semiarid Mediterranean region*, has been reached. To achieve this central objective, several research questions connecting AES, AEDS, agricultural practices, human wellbeing and agroecosystems were raised. These research questions have been indirectly answered in the three central chapters of the thesis, each group of questions referring accordingly to a respective chapter. An overview of the answers is provided here:

*Q1. (1) How can we comprehensively consider AES and AEDS?*

AES and AEDS represent the positive and negative contributions of agroecosystems to human wellbeing. This definition, although simple and direct, has many implications for the proposal of an approach for their economic valuation. The proposed approach should consider, on one side, the biophysical agroecosystem where AES and AEDS are produced, and, on the other side, the socioeconomic system where AES and AEDS are perceived and where human wellbeing should be the cornerstone of the assessment. To do so, the "Capacity, flows, demand and pressures" approach, proposed by Villamagma et al. (2013) and TEEB (2010), and readapted by Barot et al. (2017) to include disservices, was followed. This approach connects the biophysical agroecosystem to the socioeconomic system using of the AES and AEDS flows provided. Then, within the socioeconomic system, AES and AEDS are economically valued due to their impact on wellbeing. These values ultimately encompass social demand for the provision of AES and the mitigation of AEDS. In addition, this approach allows us to comprise the TEV framework, within which AES and AEDS are valued given their direct and indirect use, their option to be used, as well as their existence and opportunity to be used and enjoyed by future generations. This approach not only allows us to value the AES and AEDS socially demanded, but also the agricultural practices that may impact their provision. Agricultural practices are therefore understood as the pressures that agroecosystems receive from the socioeconomic system to modify the current provision of AES and AEDS. Agricultural practices can modify agroecosystem functioning in such a way that the actual provision of AES and AEDS is modified. However, it may also imply the introduction of additional trade-offs in their provision, which might need the assessment of their social demand. The comprehensive approach proposed here allows us to also include the valuation of agricultural practices as a way to prioritise their implementation.

*Q1. (2) What are the main AES and AEDS in semiarid Mediterranean agroecosystems? What is their relative importance?*

Following the proposed approach for agroecosystem assessment, the main ecosystem services paradigms were revised (MEA, 2005; TEEB, 2010; Haines-Young and Potschin, 2018) and a total of 12 AES and AEDS were proposed: food, water, emissions of contaminants into the atmosphere, global climate regulation, local climate regulation, water purification and waste treatment, soil maintenance, biodiversity, resilience, culture, art and design, aesthetic values, opportunities for recreation and tourism, and cognitive development and good living. A discrete choice experiment was then implemented using as attributes indicators for the AES and AEDS proposed and with the purpose of identifying the main AES and AEDS that should be economically valued in semiarid Mediterranean agroecosystems. 44 agroecosystem stakeholders were used as the target population, including farmers, agricultural researchers and policy makers.

The significance analysis of the choice experiment results allowed us to identify the main AES and AEDS that should be economically valued as well as their relative importance. Food, water, local climate regulation, water purification and waste treatment, biodiversity, opportunities for recreation and tourism were considered the main AES and AEDS to be economically valued by stakeholders. Biodiversity (38%) became the most relevant agroecosystem service to be valued, followed by recreation and tourism (20%), local climate regulation (7%), and food provision (5%). Among the AEDS, water supply for irrigation (15%) and groundwater pollution (15%) together contributed to 30% of the total importance.

*Q2. (1) What is the non-market value of each of the main AES and AEDS provided by agroecosystems in a semiarid Mediterranean region?*

Given the non-linear specification of the utility function, the marginal utility of most of the AES and AEDS is not constant. Thus, it becomes challenging to establish a unique WTP value that summarises the non-market value of each of the main AES and AEDS provided by agroecosystems in the case study area. Instead, the presence of diminishing and cross marginal effects among AES and AEDS makes their non-market value depend on their provision levels. A graphical representation of such WTP functions, therefore, becomes the best way to present their non-market value (Figure 3.3).

As such, rather than focusing on specific economic values of each AES and AEDS, it becomes more interesting to assess how these values perform in accordance with the provision levels of the different AES and AEDS, and the rationality behind these numbers. Hence, the non-market value of provisioning food decreases when the level of food provision increases. In other words, people are willing to pay more money when food security is not ensured in order to increase food provision, but when sufficient food is provided, this WTP decreases. In addition, the non-market value of food provision also depends negatively on water supplied for irrigation. The more water is supplied to provide food, the less people are willing to pay. That is, the more natural resources are consumed by agriculture, the less non-market value is attached to provided food.

Similar statements can be applied to the case of water. People positively value the use of fresh water in agriculture, and so it is reflected by their WTP. However, the more water is supplied to agriculture, the less they are willing to pay. This is so that even the non-market value of water may become negative, thereby showing WTP to reduce water supply for irrigation. Besides, the non-market value of water depends negatively on food provision and groundwater pollution. The rationality behind this fact is again clear: the more food is provided, that is, the more food security there is, the less people are willing to pay for supplying irrigation water to agroecosystems. That is, more trade-offs are perceived from using fresh water for agriculture than for environmental uses. If enough food is provided, there is no need to supply more water for irrigation, and thus the WTP decreases. In addition, people perceive the trade-offs that may arise between supplying water for irrigation and groundwater nutrient pollution. Hence, the more groundwater is polluted, the less is the value people attach to supplying water for irrigation.

Water purification and waste treatment are clearly perceived as an AEDS, and so their non-market value is always negative. This reflects the WTP people attach to the reduction of groundwater nutrient pollution, always preferring a zero level of pollution. The non-market value of this AEDS is also negatively related to water supplied for irrigation, showing again that trade-offs between both AES and AEDS are socially perceived. The more water is supplied to the agroecosystem, the more is WTP to reduce groundwater pollution. It is socially perceived that, in turn, the more likely is that higher nutrient pollution levels arise in groundwater.

In contrast, local climate regulation, genetic diversity and opportunities for recreation and tourism show constant marginal utility, thereby deriving non-market values independent of their own provision levels and other AES or AEDS.

*Q2. (2) What is the TEV of agroecosystems in this area?*

The TEV of agroecosystems in the case study area is obtained by aggregating the consumer surplus for the different AES and AEDS. This value is presented for the main agroecosystem in the case study area, summarised in terms of households and aggregated for the overall Region of Murcia considering their extension. The traditional irrigated agroecosystem provides the greatest impact on human wellbeing, which is averaged as 988 €/household/year, followed by the rainfed agroecosystem and the highly-intensive agroecosystem, whose TEV is 667 €/household/year and 501 €/household/year, respectively. On average, the wellbeing impact of AES and AEDS is valued at 650 €/household/year, which increases to 350 M€/year for the overall case study. This represents 22% of the gross added value of the agricultural sector in the Region of Murcia.

*Q3. (1) Are all the agricultural practices to mitigate nutrient pollution from agriculture equally preferred by society? Is there preference heterogeneity regarding the social demand of agricultural practices?*

The integrated valuation of AES and AEDS presented here shows that agricultural measures are more cost-effective if they are to mitigate AEDS instead of enhancing AES, as Shackleton et al. (2016) proposed. This was the case of nutrient pollution in the highly-intensive irrigated agroecosystem, where mitigating this AEDS may greatly contribute to increasing the expected wellbeing impact of such an agroecosystem. The proposed agricultural practices to mitigate nutrient pollution from agriculture were defined in Law 1/2018, on 7 February 2018, regarding urgent measures to ensure the environmental sustainability of the Mar Menor and the surrounding areas (BORM, 2018). They included (1) the banning of vegetable crops less than 100 m from the coastline (*farmland >100 m coastline*), (2) installing a system to reduce the nitrate content of the water obtained from desalination plants prior to its use in crops and its entry into groundwater and/or the Mar Menor (*denitrification plants*), (3) establishing hedgerows of native plants around farm perimeters (*perimeter hedgerow*), and (4) complying with a good agricultural practices (GAP) code based on the efficient use of fertilisers and irrigation water (*GAP code*).

The modelling results from a choice experiment covering social demand for such agricultural practices and the improvement in water quality derived from their implementation reveal the existence of two latent classes regarding social preferences. It clearly shows preference heterogeneity about the social demand of agricultural practices, and that not all the proposed practices are equally preferred. The first latent class, which represents two-thirds of the population, shows positive marginal utility for the implementation of denitrification plants, perimeter hedgerows and complying with a GAP code, in comparison with banning vegetable crops near the coastline, which was used as a baseline. Indeed, installing denitrification plants is the measure with the greatest support for this latent class. In contrast, the second latent class, which encompasses a third of the population, displays negative marginal utility for all these practices, in comparison with their baseline. This shows that for individuals in the second latent class banning on farmland less than 100 m from the coastline is the most preferred measure, or, at least, the measure that provides them with the least disutility level. Nevertheless, both classes show positive marginal utilities for the high improvement of water quality in the Mar Menor, whilst negative for the *status quo*.

The sources of preference heterogeneity that allow the distinguishing between these two latent classes are the stated environmental commitment and the perceived agricultural impact on the water quality of the Mar Menor. Thus, individuals revealing higher values of environmental commitment and who though that agriculture is mainly the responsibility of the poor water quality of the Mar Menor display a higher likelihood of belonging to latent class 2, thereby showing less support for the proposed agricultural practices.

*Q3. (2) What is the non-market value of each agricultural practice? What is the non-market value derived from the benefits of improving water quality in surrounding ecosystems?*

On average for both latent classes, the non-market value of all the agricultural practices considered is positive, which implies that, despite heterogeneity, social support is shown. As such, establishing perimeter hedgerows along farmland is the most valued measure, with a marginal WTP of 6.89 €/household/year. This is followed by compliance with a GAP code, marginally valued at 5.69 €/household/year. The establishment of denitrification plants become the least demanded measure, with a marginal WTP of 2.01 €/household/year. Moreover, the average WTP for improving water quality in surrounding ecosystems, such as the Mar Menor coastal lagoon in the case study area, is valued at 30.64 €/household/year.

## 5.2. Contributions and policy implications

As has been addressed in the previous section, the thesis objectives have been achieved and the research questions answered. Hence, it becomes natural to think that the main commitment of the thesis has been fulfilled. However, there may be some additional questions without a clear answer that would enrich the results of the present thesis. For instance, to what extent has the thesis results filled the gap in the AES and AEDS literature or non-market valuation literature? How could the results of the three central chapters be integrated? What are the policy implications of the results? Which policy recommendations could be derived from the results? In short, what are the thesis results for?

The thesis results are expected to fill a three-layered gap in the literature. The first is the theoretical gap. The results derived from the integrated non-market valuation of AES and AEDS shows that social preferences for such services and disservices follow the concavity axiom of utility functions. Most of the previous valuation from discrete choice experiments tends to assume linear utility functions, which underlines constant marginal utilities. This is an assumption not always realistic with the microeconomic theory. Another theoretical implication derived from the integrated valuation of AES and AEDS encompasses their own denomination as AES or AEDS. The presence of diminishing marginal utility for some AES and AEDS and cross effects among them have evinced the existence of quasi-AES. This new category comprises those AES that provide positive utility levels when their provision is low, but transform into disutility when their provision is high. Thus, their contribution to human wellbeing depends on their provision level, supporting the initial idea of quasi-AES proposed by Rasmussen et al. (2017).

Second, the thesis is expected to fill a practical gap. The adaptation of the main ecosystem service frameworks and classifications to the particular case of agroecosystems pretends to serve as a basis for future research in the semiarid Mediterranean area. The inclusion of AEDS in a comprehensive approach for their integrated valuation with AES also becomes a novelty that is expected to be used by researchers and practitioners. The validation of the proposed approach by agricultural stakeholders, for its part, has allowed selecting those AES and AEDS with greater importance to be economically valued. This is also to fill a practical gap for the non-market valuation of AES and AEDS in the case study area, by using stated preference methods. The integrated valuation of AES and AEDS by such these methods may become challenging when using a high number of AES and AEDS. Notwithstanding, at this stage, it is also important to note that the no consideration of some AES and AEDS by deriving their relative importance revealed by stakeholders does not mean that they do not have a value for agroecosystem assessment. It only highlights that they are less important. However, they should not be disregarded.

Third, it is the political gap. The thesis results are expected to better inform policy makers in their commitment to supporting, developing and implementing agricultural measures that result in enhancing both farmers and social wellbeing. As such, many questions come to our minds – what are the policy implications of the presented results? How can the thesis advances be applied to agricultural policy development? The results presented here are expected to be used in designing and evaluating new agricultural measures and policies. The theoretical and practical advances shown provide a wide array of political recommendations that can be employed as guidance in regional, national and even transnational agricultural policies to address the sustainability and social support of agriculture.

Non-market values should be always included in the evaluation of agricultural policies. This first policy implication is directly embedded in the overall purpose of the thesis. As it is widely known, this notion is not a novelty in the environmental and agricultural valuation literature (Sandhu et al., 2008; De Groot et al., 2012; Costanza et al., 2017). However, never hurts to remember it again, particularly, when the focus is on human wellbeing as in the case of AES and AEDS. On the one hand, focusing merely on market values involves considering only food provision in the case of agroecosystems, which might not be the most important AES for stakeholders and society, as revealed, in particular when high levels of food provision are ensured. Therefore, expanding the frame of valuation to encompass non-market values implies also the consideration of a broad range of AES and AEDS, embracing agriculture's multifunctional character. On the other hand, non-market valuation allows human wellbeing to become the foundation of economic values, mainly when demand-side approaches are taken into account, as is the case shown here. Most of the AES and AEDS take the form of public goods or externalities so including their non-market value in the evaluation of policies becomes the only way to ensure they are taken into account.

AEDS should be included in agricultural policy evaluation. Overlooking AEDS means disregarding the expected negative contributions of agricultural policies to human wellbeing. As it has been shown throughout this thesis, this could lead to incorrect policy decisions. First, because the negative contributions of AEDS undermine the positive outcomes of AES. Even, in an extreme, they could be greater, advocating into negative net contributions of agroecosystems. Perhaps better solutions would be those which show lower provision levels of AES, in addition to lower provision levels of AEDS. Thus, not considering AEDS could make policy-makers obfuscated by merely looking at the positive contributions of AES. Second, developing new agricultural policies could be more cost-effective and have a higher impact on human wellbeing by reducing AEDS than by increasing AES. This is what the results presented here show. Reducing groundwater pollution or irrigation water supply provides a higher positive impact on wellbeing than increasing the provision of AES in the highly-intensive irrigated agroecosystem. Hence, policy-makers are encouraged to be aware of the importance of AEDS when designing and evaluating policies. It implies not only taking into account AEDS with the focus on their mitigation but also considering the trade-offs that may arise between them and AES when policies are implemented. In consequence, disregarding the value of AEDS when evaluating agricultural policies may not only compromise the expected wellbeing impacts of such policies, but also the sustainability of agriculture. Economic valuation of AES and AEDS allows us to convert into monetary terms the economic, social, and environmental impacts of agriculture, therefore enabling the application of traditional economic tools to the evaluation of policies, such as the cost-benefit analysis.

In addition, the integrated consideration of AES and AEDS for their economic valuation allows us to take a step forward in the valuation of agricultural contributions to human wellbeing. To date, most of the negative contributions of agriculture were considered as externalities or “bad” goods – as opposed to public goods – (Villanueva et al., 2018). Similarly, in the case of AES, they were valued depending on their private or public nature. This makes the integration of the different economic values under a common approach challenging. The ecosystem service framework, which allows us to value all the contributions of human wellbeing irrespective of their consideration as private or public goods or externalities, embraces market and non-market valuation to ultimately obtain the economic value of agroecosystems and any proposed agriculture policy that may change the provision of AES and AEDS. In such a context, the present thesis is just one of the first attempts to show the importance of considering AES and AEDS in agroecosystem assessment and valuation in an integrated way, to show a comprehensive approach to perform that, and to display the potentialities of using such an approach.

Looking into the details, the thesis results have allowed us to understand which AES and AEDS have a greater impact on human wellbeing, and how they could be managed to increase agriculture's contribution to wellbeing. Specific guidelines for the design of ongoing agricultural policies in view of enhancing human wellbeing are proposed. First, agricultural policies that enhance food provision are encouraged but they should not be the main focus. Food security become the AES that provides the greatest contribution to human wellbeing, according to values revealed by its social demand. However, when relatively high levels of food security are ensured, as is the case in most of the semiarid Mediterranean region, food provision does not need to be the only priority for policy.

Second, marginal irrigation water productivity should be maximised. As stated throughout the thesis, supplying water for irrigation is socially demanded. However, it is also socially claimed that not all the available fresh-water is used for irrigation purposes, as diminishing marginal utility and negative WTP for water supply for irrigation at high levels was revealed. The adoption of new agricultural practices by farmers that minimise irrigation water consumption might be therefore a priority for policy-makers, not only because of the ongoing decrease of fresh-water resources due to climate change – *this is another issue* –, but also because wellbeing depends on it. Regulated deficit irrigation strategies could be, for instance, one of the agricultural practices to promote. Besides, the use of alternative sources of irrigation water, such as reclaimed or desalinated water, are also welcomed to support such strategies. This is widely implemented in the case study area (Martínez-Alvarez et al., 2017), and the results shown here seek to support this common practice in the area. Hence, agricultural water management can benefit from the findings here obtained. Actually, they were employed by Alcon et al. (2022) to untangle the total economic value of irrigation water in the semiarid Mediterranean region. They proposed alternative scenarios to address agricultural water in the context of climate change and maximise the different agricultural contributions to human wellbeing.

Third, the promotion of biodiversity-friendly environments is encouraged. Society demands that agriculture respects and enhances biodiversity in agricultural landscapes. Widely extended monocultures, mainly based on the intensive use of chemicals (pesticides, herbicides, fungicides, fertilisers, etc.), has evoked a depletion of agriculture-related biodiversity (Beckmann et al., 2019). This situation is due to be reverted and, its change is socially demanded. Farmers are therefore under pressure to adopt the new agricultural practices. Reducing the use of agricultural pesticides, encouraging pests' natural enemies and biological control of plagues is plausible nowadays. For instance, crop diversification strategies, cover crops, or farm perimeter hedgerows are agricultural practices that have demonstrated the ability to overcome such challenges while at the same time promoting agricultural biodiversity (Rosa-Schleich et al., 2019). Policy makers should commit to promoting such practices among farmers to ensure their adoption and implementation. The results here provided could be used as a way to support the launch of economic instruments, such as farm subsidies or tax reductions, to stimulate farmers' adoption of biodiversity-friendly practices.

Fourth, agroecosystems should be understood not only as AES and AEDS production systems but also as complex socio-ecological systems where all the spheres of wellbeing are addressed and promoted. This links directly to the social demand for opportunities for recreation and tourism in the agroecosystems. Simple actions, such as allowing little paths for walking or cycling around farm plots, would be enough to promote the social enjoyment of the agroecosystems. However, this kind of initiative is partly developed within the case study area. Only rainfed and traditionally irrigated agroecosystems are allowed to have such recreation activities. This recreational gap within highly-intensive irrigated agroecosystems might be the incentive for developing new policy actions that ensure that such AES is provided by these irrigated agroecosystems. This would become an additional way of increasing the wellbeing contributions of highly-intensive irrigated agroecosystems in the semiarid Mediterranean area.

Fifth, but not least, mitigating AEDS from agroecosystems should be a priority for policy-making, especially in those agroecosystems whose provision is significantly high. This is particularly the case in highly-intensive irrigated agroecosystems in the case study area, which may impact upon surrounding water ecosystem through nutrient pollution, becoming a prime concern for agri-environmental policy (MITERD, 2021). Society claims policy actions to mitigate such pollution, which highly affects surrounding ecosystems, and whose non-market value summarises such a negative impact on wellbeing. Banning vegetable crops near the coastline, the installation of denitrification plants to reduce the nitrate content of agricultural wastewater, establishing perimeter hedgerows around farm plots and the compliance of a GAP code are examples of agricultural measures to mitigate such pollution. However, the range of feasible agricultural practices that can be promoted by agricultural policy-makers is broader. For instance, the use of green manure, crop diversification with legumes, or incorporating crop residues into the soil are also good agricultural practices that positively impact soil fertility and reduce fertiliser needs (Sánchez-Navarro et al., 2019). Regardless of the specific agricultural practices to be promoted, the thesis results have brought up the need for agricultural actions to address AEDS as well as the importance of socially assessing such measures they are finally implemented by policy-makers. The acceptance and public support of policy decisions are key even in agricultural policy. The findings of the present study have served to raise awareness about the significance of preference heterogeneity assessments, even in agricultural policies, and from them designing political strategies and incentive systems that ensure the ex-ante acceptance and support of future measures.

In sum, agricultural policies, such as the Common Agricultural Policy (CAP), would benefit from the integrated valuation of AES and AEDS, concerning the adjustment of regional payments according to their social benefits and the design of potential eco-schemes to be included in the future CAP, to reach the Green Deal targets. Therefore, this work is expected to be useful for the making of decisions about agricultural management, with a special focus on the semiarid wester Mediterranean region, and for the adoption of agricultural policies that maximise the social wellbeing that the agroecosystem imparts to society, by showing sustainability criteria, including the environmental and social costs. It will also support agricultural policies in the establishment of normative criteria and the design of economic instruments in water-scarce areas by assessing environmental sustainability and providing a reference for compensation.

## 5.3. Opportunities for future research

The integrated non-market valuation of AES and AEDS by using stated preference methods is a novel topic in agri-environmental literature. makes this research topic an interesting matter to deeply address in future research. To delve into the practical implications and applications of the results, to verify the microeconomic basics of discrete choice experiments, and to expand the assessment to the farmers' viewpoint, are brief examples of the range of opportunities that may arise from the current research. Some of these opportunities for future research will therefore be proposed and discussed.

Choice experiments are assumed to follow the microeconomic basics regarding individual preferences. Thus, individual preferences comply with the axioms of completeness, transitivity and monotony, which can be expanded to encompass the continuity and concavity of utility functions. As it has been revealed, some of these basics have been demonstrated in the thesis, in particular the continuity and concavity of utility functions. However, there is still room for research about the verification of the rest of the axioms, in particular in the case of integrated AES and AEDS valuation where literature covering this topic is scarce. Furthermore, the economic value derived from the employment of choice experiments are assumed to be true values, that is, the real economic value that reflects the individual needs that are satisfied. Notwithstanding, this becomes challenging to address. In such a context, checking the validity and reliability of the obtained results provides a great opportunity for further research. This becomes even more interesting since the choice experiment results here presented comprehend negative and positive effects for attributes. Understanding to what extent the economic values here derived comply with the basics of preference theory will help to enhance trust in the use of such approaches for deriving economic values for policy evaluation. Despite the challenges it may involve, using alternative non-market valuation techniques or a test-retest approach to ensure the reproducibility of the choice experiment results are also additional opportunities for future research, and where the reliability of the employed method will be checked.

In line with the previous comment, disentangling the sources of preference heterogeneity about the social demand for AES and AEDS helps us to better understand the sociodemographic, attitudinal and behavioural factors that determine the value individuals attach to them. This would allow us to orientate policy actions, not only to the supply of AES and AEDS to satisfy social demand as proposed before, but also to shape social demand in favour of a higher impact of such policies in human wellbeing.

Only non-market values have been estimated for AES and AEDS within the present thesis. However, as it is known, the agroecosystem contributions also encompass goods and services which are valued through markets. This is important to have in mind when considering the policy implications of the results, mainly because any policy evaluation should include both market and non-market values before taking any decision. Therefore, integrating the non-market value of AES and AEDS here obtained with their respective market valuation provides an opportunity for further research to encompass the overall value of agroecosystems. Moreover, expanding the economic valuation to include all the possible AES and AEDS, and not only those which were found most significant to explain stakeholders' choices, represents another opportunity for further research. This is actually what Alcon et al. (2022) showed. The results obtained within this thesis were broadened and integrated with the market and non-market values for the rest of AES and AEDS to value the contribution of irrigation water to traditional and highly-intensive irrigated agroecosystems. Then, the authors used market valuation for food, benefit transfer for the non-market value of global climate regulation, soil maintenance, cultural heritage and aesthetic landscape values, as well as shadow prices for employment generation.

The results here obtained seek to be the basis for a better design of agricultural policies that will enhance human wellbeing. As discussed, this firstly involves the design of new agricultural practices that improve the provision of AES and mitigate AEDS. Practices that enhance food provision, biodiversity, temperature regulation and opportunities for recreation in irrigated agroecosystems are encouraged, whilst minimising the consumption of irrigation fresh water and nutrient pollution. The presence of trade-offs among AES and AEDS may even arise when agricultural practices put pressure on agroecosystems. The opportunity for further research requires now not only understanding of the social demand for AES and AEDS, but also agronomic and ecological knowledge to optimise the agroecosystem impacts and contributions when proposing such agricultural practices.

Social demand is the main focus of the thesis. For the purpose of the research, the provision of AES and AEDS has been mainly considered as exogenous, coming from farmers' decisions that impact human wellbeing. Farmers' viewpoints and preferences have not been included in the frame of assessment. Indeed, this is not needed when the purpose is merely to value the contributions of agroecosystems to human wellbeing. However, the situation changes when we seek to propose changes in current agricultural practices to embrace social demand for AES and AEDS. Are farmers willing to provide the socially demanded AES and mitigate the current provision of AEDS? To what extent are farmers willing to accept these new agricultural practices? What are the incentives that may ensure farmers accept them? Addressing the answers to these questions provide an interesting matter for future research. The supply-side of AES and AEDS has a key role in the agroecosystem contributions to human wellbeing and thus it should be reflected in the future agri-environmental research.

## 5.4. References

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# Appendix. Scientific quality of the publications conforming the PhD thesis

## A.1. Article 1

Zabala, J.A., Martínez-Paz, J.M., Alcon, F., 2021. A comprehensive approach for agroecosystem services and disservices valuation. Science of The Total Environment 768, 144859. https://doi.org/10.1016/j.scitotenv.2020.144859. Q1. IF (2020[8]) 7.963.

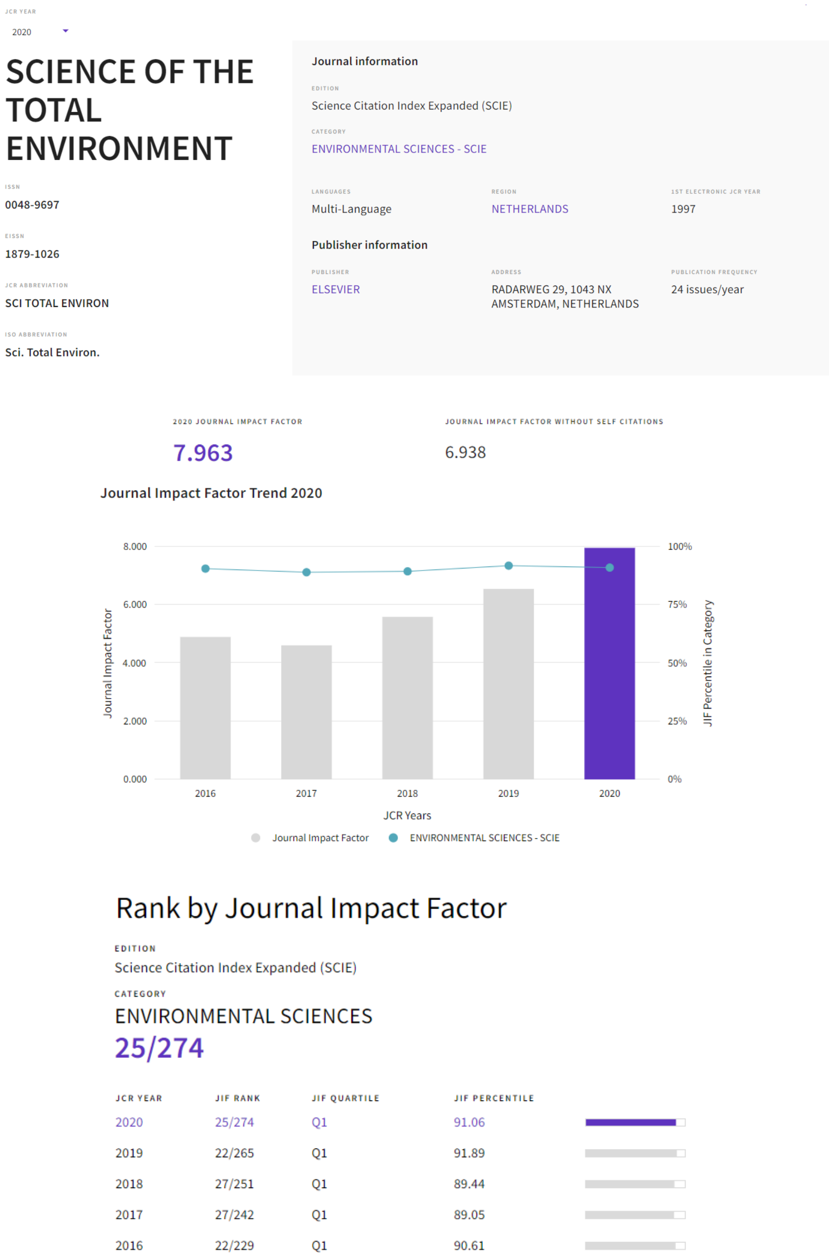


Figure A.1.1. Journal Impact Factor metrics of Article 1

[8] Last year available

## A.2. Article 2

Zabala, J.A, Martínez-Paz, J.M., Alcon, F., 2021. Integrated valuation of semiarid Mediterranean agroecosystem services and disservices. Ecological Economics 184, 107008. https://doi.org/10.1016/j.ecolecon.2021.107008. Q1. IF (2020[9]) 5.389

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2020

**ECOLOGICAL ECONOMICS**

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EDITION
Social Sciences Citation Index (SSCI) | Science Citation Index Expanded (SCIE)

CATEGORY
ENVIRONMENTAL STUDIES - SSCI | ENVIRONMENTAL SCIENCES - SCIE
ECONOMICS - SSCI | ECOLOGY - SCIE

LANGUAGES: English | REGION: NETHERLANDS | 1ST ELECTRONIC JCR YEAR: 1997

**Publisher information**

PUBLISHER: ELSEVIER | ADDRESS: RADARWEG 29, 1043 NX AMSTERDAM, NETHERLANDS | PUBLICATION FREQUENCY: 12 issues/year

2020 JOURNAL IMPACT FACTOR
5.389

JOURNAL IMPACT FACTOR WITHOUT SELF CITATIONS
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Rank by Journal Impact Factor

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| JCR YEAR | JIF RANK | JIF QUARTILE | JIF PERCENTILE |
|---|---|---|---|
| 2020 | 27/376 | Q1 | 92.95 |
| 2019 | 26/373 | Q1 | 93.16 |
| 2018 | 20/363 | Q1 | 94.63 |
| 2017 | 21/353 | Q1 | 94.19 |
| 2016 | 28/347 | Q1 | 92.07 |

Figure A.1.2. Journal Impact Factor metrics of Article 2

[9] Last year available

## A.3. Article 3

Alcon, F., Zabala, J.A., Martínez-Paz, J.M., 2022. Assessment of social demand heterogeneity to inform agricultural diffuse pollution mitigation policies. Ecological Economics 191, 107216. https://doi.org/10.1016/j.ecolecon.2021.107216. Q1. IF (2020[10]) 5.389.

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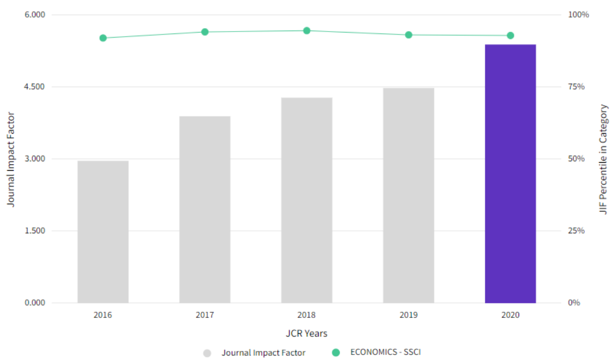


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|---|---|---|---|
| 2020 | 27/376 | Q1 | 92.95 |
| 2019 | 26/373 | Q1 | 93.16 |
| 2018 | 20/363 | Q1 | 94.63 |
| 2017 | 21/353 | Q1 | 94.19 |
| 2016 | 28/347 | Q1 | 92.07 |

Figure A.1.3. Journal Impact Factor metrics of Article 3

[10] Last year available